Developing Ecological Criteria for Prescribed Fire in South Florida Pine Rockland Ecosystems

(USGS Open File Report OF 2006-1062)

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Executive Summary

The pine rocklands of South Florida, characterized by a rich herbaceous flora with many narrowly endemic taxa beneath an overstory of south Florida slash pine (*Pinus elliottii* var. *densa*), are found in three areas: the Miami Rock Ridge of southeastern peninsular Florida, the Lower Florida Keys, and slightly elevated portions of the southern Big Cypress National Preserve. Fire is an important element in these ecosystems, since in its absence the pine canopy is likely to be replaced by dense hardwoods, resulting in loss of the characteristic pineland herb flora. Prescribed fire has been used in Florida Keys pine forests since the creation of the National Key Deer Refuge (NKDR), with the primary aim of reducing fuels. Because fire can also be an effective tool in shaping ecological communities, we conducted a 4-year research study which explored a range of fire management options in NKDR. The intent of the study was to provide the Fish and Wildlife Service and other land managers with information regarding when and where to burn in order to perpetuate these unique forests.

In 1998 we initiated a burning experiment in a randomized complete block design. Three treatments were to be carried out in a single well-defined block in each of two characteristic understory types during each year from 1998 through 2000. One understory type was characterized by a relatively sparse shrub layer and a well-developed herb layer ("open"), and the second had a dense shrub layer and poorly developed herb layer ("shrubby"). The three burn treatments were: (a) summer burn, (b) winter burn, and (c) no burn, or control. Three 1- ha plots were established in each block, and randomly assigned to the three treatments. Though the first year experimental burns were carried out without incident, constraints posed by external factors, including nationwide and statewide prohibitions on prescribed burning due to wildfires in other regions, delayed the experimental burns and precluded collection of postburn data on one third of the burns. Ultimately we burned only eleven plots, three in winter and eight in summer, over a four-year period from 1998 to 2001. Vegetation was sampled in a stratified, nested design within 18 plots. Trees were sampled in a 1.0-ha plot, shrubs in twenty 50-m^2 circular (radius 4 m) subplots within the tree plot, and the herb layer in four circular 1-m^2 quadrats (radius 0.57 m) within each subplot. The amount of fuel in the shrub layer was estimated by applying regression models to plant dimensional data, and ground layer fuel was estimated by a harvest method. The effects of Key deer herbivory on regeneration of the understory pine rockland plant community after fire was studied by monitoring inside and outside exclosures established within two of the six blocks.

Pine trees constituted more than half (53.3%) of the biomass, but understory fine fuels comprised a surprisingly high proportion of total aboveground biomass. In the three blocks in which paired summer and winter burns were successfully conducted, the summer burns were more intense than the winter burns as judged by our indicators of fire intensity. Because of the differences in fire intensity between seasons, it was not possible to say whether observed differences in vegetation response between summer and winter burns were due to season or to fire intensity. The mortality of South Florida slash pine trees was greater after the summer burn than the winter burn in each block, but other vegetation responses were rarely as consistent. For instance, *Metopium* showed less recovery after summer burns in two blocks and after the winter burn in the third block. Moreover, there were instances in which alternative growth stages of the same species responded differently. Adult palms succumbed more frequently to summer than winter burns, and mortality of *Coccothrina*x exceeded that of *Thrinax*. In contrast, small palms recovered more readily after summer burns than winter burns. High intensity fires may be beneficial for some plant responses, e.g., the germination of *Jacquemontia pentantha* seeds. Deer herbivory was greater in burned areas, but also present on some plants in unburned control areas.

During the course of this study a series of permanently marked vegetation plots were established. There are now 18 permanently marked 1.0-ha plots on Big Pine Key in which all trees with DBH ≥ 5 cm (> 11,400 stems) have been tagged and measured. Plots should be periodically re-measured to follow growth, mortality, and recruitment into the tree stratum. Within each of the 18 plots there are also 20 marked shrub plots in which woody plants and palms ≥ 1 m tall have been tallied, and 80 individually marked permanent herb quadrats in which the abundance of all species present has been estimated. These plots will be immensely valuable to document future changes, whether due to prescribed fire, wildfire, hurricane, sea level rise, or other unforeseen disturbances.

1. INTRODUCTION

Background

The pine rocklands of South Florida are globally endangered ecosystems containing a rich herbaceous flora with many narrowly endemic taxa beneath an overstory of south Florida slash pine (*Pinus elliottii* var. *densa*). These once-extensive communities have borne more than their share of the development that has occurred in the region, and are now greatly reduced in area and divided into a number of smaller fragments (U.S. Fish and Wildlife Service 1999). The increasing influence of man in the landscape has altered many critical ecosystem processes, including the fire regime, hydrological balance, availability of exotic seed sources, and extent of gene flow within many key species. It is critical that a management strategy for conserving South Florida rocklands be developed to address these threats.

Pine rocklands are found in three areas of South Florida (Snyder et al. 1990): the Miami Rock Ridge of southeastern peninsular Florida, with the largest remaining area in Everglades National Park; the Lower Florida Keys, mostly within the National Key Deer Refuge; and the southern Big Cypress pinelands, entirely within Big Cypress National Preserve. The latter pine rocklands are perhaps better considered transitional to typical pine flatwoods to the north because a shallow layer of sand generally covers the limestone substrate and many of the tropical elements of the flora are missing. The Miami Rock Ridge and the Lower Keys pinelands grow on a substrate of oolitic limestone. They are characterized by a very diverse shrub layer dominated by hardwoods of West Indian distribution and a few species of palms. The herb layer is likewise diverse and contains 35 taxa endemic to southern Florida, including several species listed or under review for listing by the federal government (Snyder et al. 1990). Similar pine rocklands exist in several of the Bahama islands, although the canopy tree in this case is *Pinus caribaea* var. *bahamensis*.

The importance of fire as a part of this ecosystem has long been recognized (Robertson 1953, Wade et al. 1980). In the absence of fire, a closed hardwood canopy develops and the characteristic pineland herb flora is lost (Robertson 1953, Alexander 1967, Loope and Dunevitz 1981). In fact, it is the requirement of these endemic herbaceous species for fire that helped lead to the establishment of the prescribed burning program in Everglades National Park over 50 years ago (Robertson 1953). In the Lower Keys pinelands, the importance of fire for Key deer habitat maintenance is well known (Dickson 1955, Alexander and Dickson 1972, Carlson 1989). Robertson (1955) studied the vegetation of the area as part of an analysis of South Florida breeding birds. Recent studies by Ross et al. (1992a,b; 1994) document the physical factors associated with the distribution of Lower Keys pinelands.

Lower Keys pine rockland

The pine forests of the Lower Keys are a small but biologically important representative of South Florida pine rocklands. There are less than 1000 ha of Lower Keys pinelands scattered over seven islands, yet this ecosystem is the major habitat for the Key deer (Dickson 1955). It is also home to 14 herbs endemic to South Florida, five of which occur only in these Lower Keys settings (Avery and Loope 1980). In addition to several endemic plants under consideration for federal listing (e.g. *Chamaecrista lineata* var. *keyensis*, *Chamaesyce deltoidea* var. *serpyllum*, *Linum arenicola*, *Melanthera parvifolia* [Ross and Ruiz 1996]), two endemic butterfly taxa, the Florida leafwing and Bartram's hairstreak, are under consideration for listing (Wood 1994). The larval food plant of both butterflies is *Croton linearis*, an herb common in Keys pinelands.

Protection and management of the Lower Keys pinelands is made particularly difficult by the pattern of land ownership and development. Much of the remaining pinelands are in small, privately owned parcels and there are private inholdings within the National Key Deer Refuge. Private residences embedded within the fire-sustained pineland habitat represent a classic example of the urban/wildland interface problem. In a more fragmented but otherwise similar setting in southern Miami-Dade County, an *Ips* beetle outbreak after Hurricane Andrew has left many of the isolated parcels without a single mature pine tree. The severity of the insect damage has been attributed to improper fire management by several researchers (Doren et al. 1993; Loope et al. 1994). Great expense and tenacity will be required to salvage even a remnant of the urban pine rockland ecosystem in Miami under current conditions. The daunting problems faced by resource management agencies include severe invasion by exotic trees and grasses, widely scattered sources of pine regeneration, and little experience with prescribed fire in such close quarters. The situation in the Keys is not so bleak, and the results of this study should assist in restoration of the Miami-Dade County pinelands.

There has been sporadic prescribed burning in the National Key Deer Refuge since its creation (Berg and Wisby 1996). From 1985 to 1992 prescribed burns were conducted in the Refuge mainly for fuel reduction, without much consideration of ecological factors such as season of burning. It has been shown elsewhere that season of burning can have marked influence on the response of pines, understory hardwoods, and herbaceous plants (Robbins and Myers 1992, Platt et al. 1988, Snyder 1986, Spier and Snyder 1998, Waldrop et al. 1992). There was no prescribed burning by U.S. Fish and Wildlife Service (FWS) staff in the Refuge from 1992-1997, in part because not enough was known about the ecological effects of prescribed fire in this system.

Objective

The objective of this study was to document the response of pine rockland vegetation to a range of fire management options. The ultimate goal of the project was to provide FWS and other land managers with information useful in deciding when and where to burn to perpetuate these unique pine forests.

Study design

Our proposal called for experimental burns to be carried out at two times of the year: during the rainy season, when lightning-caused fires are likely to occur (Robertson 1953, Wade et al. 1980, Snyder 1991) and when plants are actively growing; and in the dry season, when plants are most dormant and burning conditions are relatively mild. For simplicity, the two seasonal treatments are referred to as summer and winter, respectively. In addition to comparing season of burning, we wanted to compare burns in two contrasting types of pineland: pineland with relatively sparse shrub layer and a well-developed herb layer, referred to as open, and pineland with a dense shrub layer and a less developed herb layer, referred to as shrubby.

On Big Pine Key we identified three blocks of open pineland and three blocks of shrubby pineland. Within each block we established three 1.0-ha experimental plots, one to be burned in the summer, one to be burned in the winter, and one to remain unburned as a control (Table 1.1, Figure 1.1). Each year for three years the treatment burns were to be applied to the units in one open and one shrubby block. There were therefore a total of 18 experimental units: 2 types of pineland (open and shrubby) X 3 treatments (summer burn, winter burn, and unburned) X 3 years. Postburn data were to be collected annually so that the first set of burns would have three years of postburn data, the second set would have two years of postburn data, and the last year's burns would be followed for one year.

The experimental plots were 100 x 100 m in most cases, but at the Poisonwood site they were 80 x 125 m and at the Buttonwood site they were 50 x 200 m in order to fit within available habitat. GPS coordinates of plot corners are given in Appendix 1. Plots were laid out with tapes and right angle prisms. Shrubs and herbs were sampled in circular subplots within the 1.0-ha plot. To locate the understory sampling points, the plots were divided into a 10 m by 10 m grid with a buffer 10 m wide around the periphery of the plot. Inside the buffer, potential sampling areas formed a checkerboard pattern so that subplots would not share a common boundary. Twenty sampling points were randomly chosen from the available points: 32 potential points in square plots, 30 points in 80 x 125 m plots, and 27 points in 50 x 200 m plots (See Figure 1.2). A short piece of steel reinforcement bar (rebar) was driven into the limestone in the center of each of these 10 m by 10 m sampling areas. Shrubs were sampled in circular 50-m^2 plots (radius 4 m) centered on the rebar in each of the twenty subplots, referred to as "shrub plots." The herb layer was sampled in four circular 1-m^2 plots (radius 0.57m) centered 2 m from the shrub plot center in the four cardinal directions. Therefore there were 80 "herb plots" in each of the 1.0-ha experimental units. Details of sampling methods are contained in subsequent sections of the report. Scientific names and authorities of all vascular plant species found in the study plots are found in Appendix 2.

The experimental burns were carried out by U.S. Fish and Wildlife personnel under the direction of James E. Durrwachter, FWS Fire Management Officer for South Florida, with assistance provided by The Nature Conservancy (TNC) through their fire management cooperative agreement with FWS.

The first year the experimental burns were carried out without incident—two summer burns in August and two winter burns in December. Hurricane Georges made landfall during mid-morning of September 25 in Key West with a minimum central pressure of 981 mb and maximum winds of 90 knots. Fortunately, the hurricane did not result in substantial wind damage and effects of saltwater from storm surge did not reach any of the study sites. The two summer burns the following year, 1999, also were done on schedule, but, from then on, circumstances beyond our control prevented us from implementing the study design as proposed.

In late summer and fall of 1999, extensive fire activity in Nevada and California shut down prescribed burning nationwide, and by the time burning was authorized that winter, it was judged too dry to burn on Big Pine Key. In May 2000, prescribed burning was dealt a serious blow by the repercussions of an escaped prescribed fire in Bandelier National Monument, New Mexico. The Cerro Grande fire resulted in a nationwide prescribed fire moratorium. We could not do the summer burns that year and we were fortunate to get one of the postponed winter burns from 1999 completed in December 2000. This put us effectively one year behind schedule in getting the treatment burns done. Because the project funding was due to run out at the end of 2001, we opted to burn all four of the plots in the last two study areas in the summer of 2001. However, this was done with the understanding that we would not be able to collect the one year postburn data on the plots the following year.

Table 1.1. Original design of seasonal burning treatments.

Site	Pineland type	Year	Treatment	Plot code
Orchid	Open	One		
			Control	OC
			Summer burn	OS
			Winter burn	OW
Poisonwood	Shrubby			
			Control	PC
			Summer burn	PS
			Winter burn	PW
Iris	Open	Two		
			Control	IC
			Summer burn	IS
			Winter burn	IW
Dogwood	Shrubby			
			Control	DC
			Summer burn	DS
			Winter burn	DW
Locustberry	Open	Three		
			Control	LC
			Summer burn	LS
			Winter burn	LW
Buttonwood	Shrubby			
			Control	BC
			Summer burn	BS
			Winter burn	BW

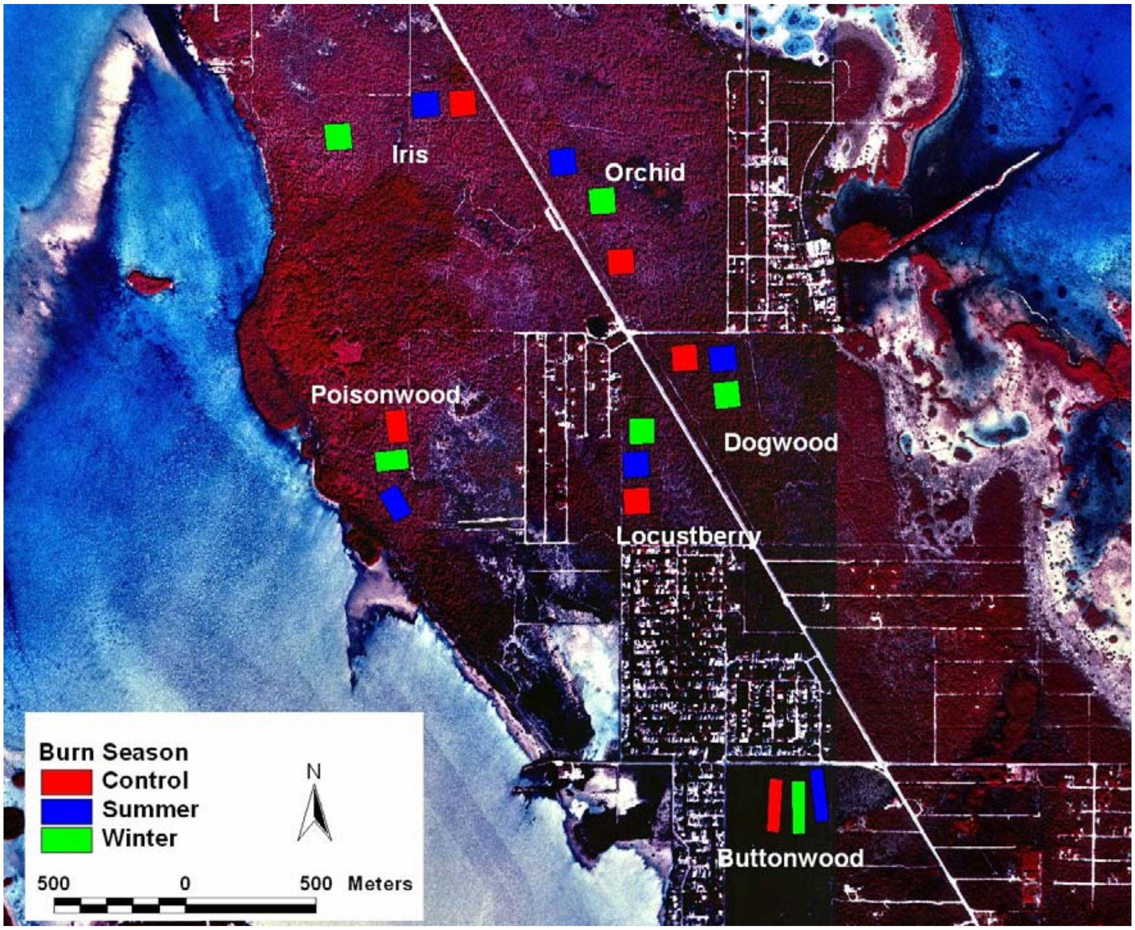

Figure 1.1. Map showing location the 18 experimental plots at six sites on Big Pine Key. GPS coordinates of plot corners are listed in Appendix 1.

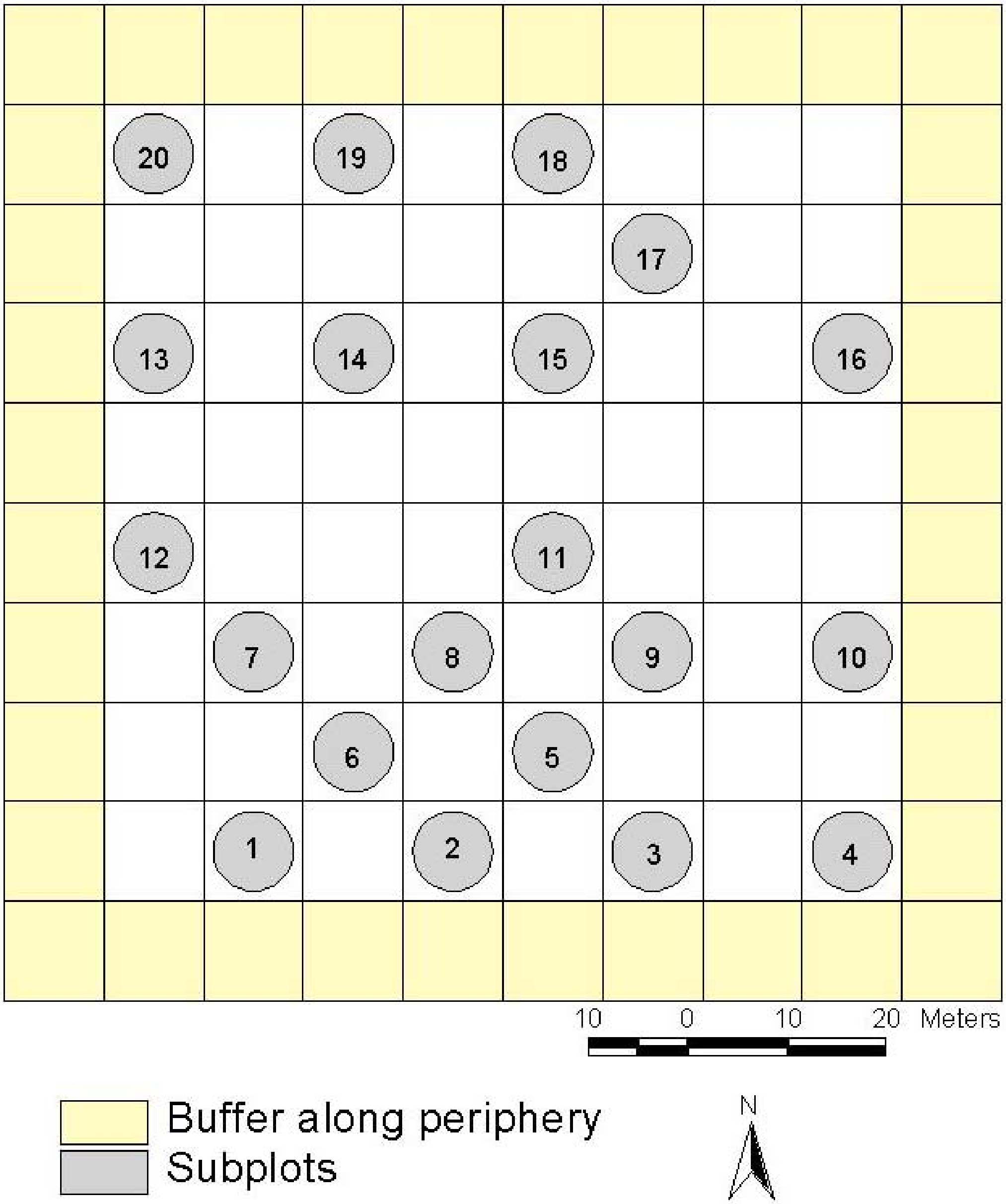

Figure 1.2a. Layout of 100x100 m plot showing 10 m buffer and location of 20 understory sampling locations. Used at Orchid, Iris, Dogwood, and Locustberry sites.

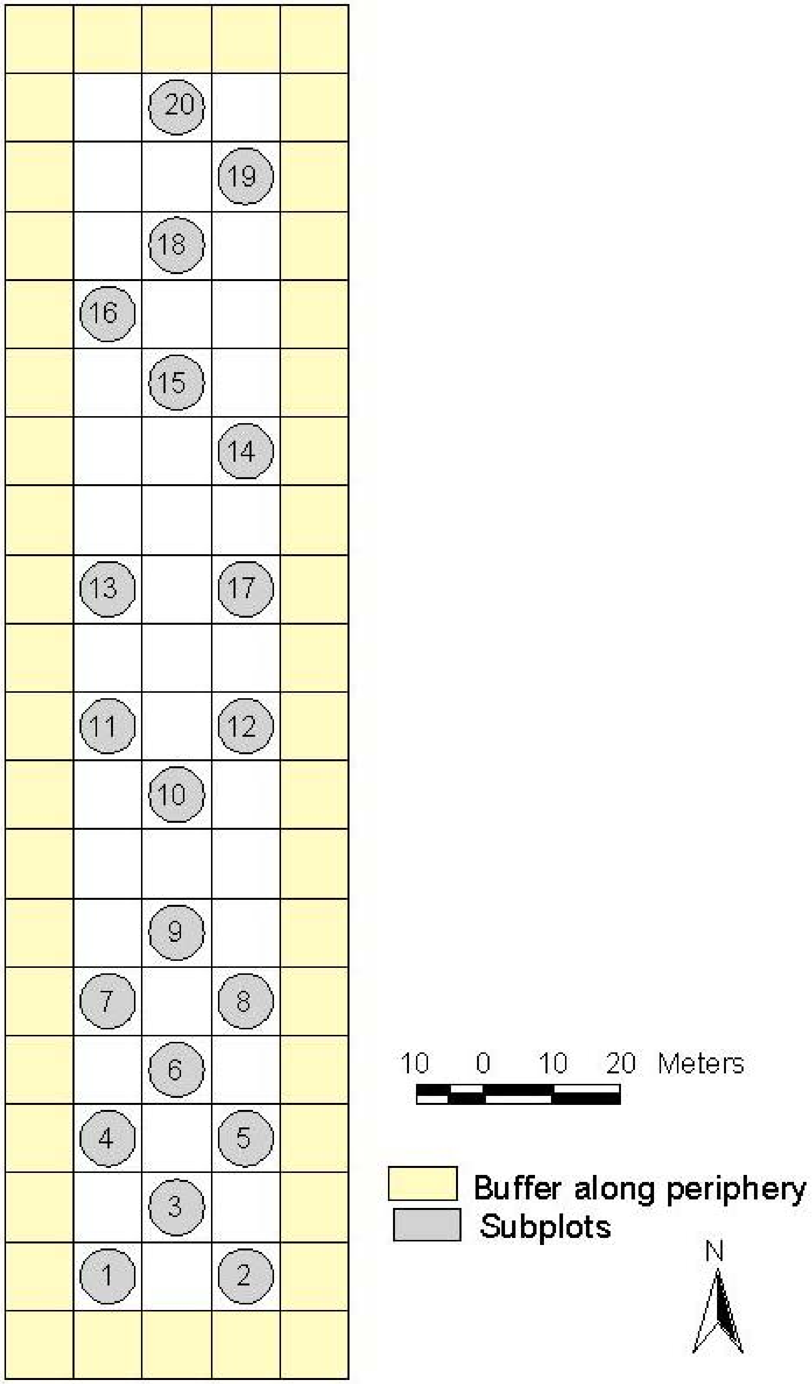

Figure 1.2b. Layout of 50x200 m plot showing 10 m buffer and location of 20 understory sampling locations. Used at Buttonwood site.

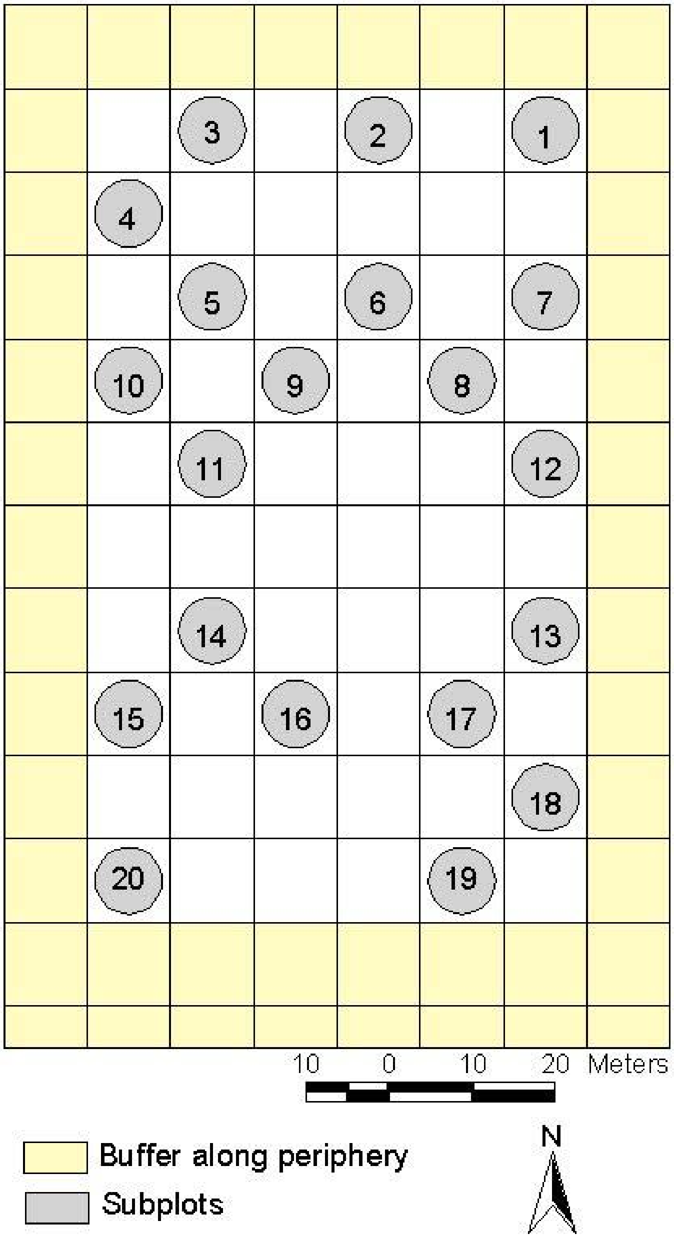

Figure 1.2c. Layout of 80x125 m plot showing 10 m buffer and location of 20 understory sampling locations. Used at Poisonwood site. The winter burn plot was rotated 90° in a clockwise direction to run east-west.

2. EXPERIMENTAL BURNS

Methods

The selection of study areas was done in conjunction with the FWS staff in order to assure that the experimental burns could be carried out safely. When necessary, firebreaks were cut around the plots to prevent the fires from spreading beyond the intended target. The prescribed burns were carried out by FWS with assistance from TNC.

We recorded four indicators of fire intensity: fire temperature, fuel consumption, and stem char height and canopy scorch of pine trees. During the first summer burns (Orchid and Poisonwood) we wrapped sets of 16 temperature pellets that melted at temperatures of 150 to 900° F (in increments of 50° F) in heavy duty aluminum foil. One set of pellets was placed on the ground in each of the 20 shrub plots. Unfortunately, there was some difficulty in interpreting the results (pellets blackened but not melted were recorded as melted) so we switched to using steel plates with temperature-sensitive paints for the seven burns done the final 3 years of the project. Ten paints that melted at temperatures ranging from 200 to 650^{o} F in 50^{o} F increments (Tempilaq, Big Three Industries) were dabbed on 7.5 cm x 7.5 cm x 3 mm steel plates. The plates had a small hole drilled in one corner for hanging on a wire support. The lowest temperature paint was placed at the bottom of the plate and the highest temperature was placed at the top so that melting paints did not obscure higher temperature paints. One plate was placed at ground level in each herb plot for a total of 80 per burn.

Fuel loads were measured in 0.25-m^2 (0.5 x 0.5 m) clipped quadrats. Before burning, quadrats were located 4.5 m from the center of each shrub plot in the four cardinal directions. After burning, the adjacent area in a clockwise direction was sampled for the remaining fuel. We collected all live and dead plant material from the ground surface up to 1 m above ground level. The upper size limit for dead material was 2.5 cm in diameter and for live material was 0.5 cm diameter. The biomass from the quadrats at each shrub plot was combined and sorted into the following classes: forbs, ferns, graminoids, woody plants, palms, litter < 0.5 cm diameter, and litter ≥ 0.5 cm and < 2.5 cm. The material was dried to constant weight in a drying oven set to 70° C. Fuel consumption was estimated by subtracting the post-burn fuel weight from the pre-burn fuel weight. The post-burn fuel samples were not collected after the Iris winter burn.

Crown scorch percent and char height, two commonly used surrogates of fire intensity (Menges & Deyrup 2001), were measured in all burned plots, and were used as estimators of intensity. For each pine tree, the height of charred bark and the percentage of the crown scorched were recorded within a month of the burn. Maximum char height was directly measured on individual trunks, with directional orientation noted (8 possible compass directions: N, NE, E, SE, S, SW, W, NW). Percent scorch, a measure of the amount of live needles killed by the fire (percent crown scorch volume, Fowler and Sieg 2004), was estimated to the nearest 5 or 10%.

Results and Discussion

A total of 11 of the planned 12 experimental burns were carried out (Table 2.1). In 1998, the summer and winter burns in Orchid and Poisonwood blocks burned on schedule. In 1999, only the summer burns were completed in the Iris and Dogwood blocks. The paired winter burn for the Iris block was completed a year late, in December 2000 and the Dogwood winter burn plot was never burned. In the summer of 2001, the summer and winter burn plots in the Locustberry and Buttonwood sites were burned.

The burns were carried out within a few days of a significant rain event with relatively low in-stand wind speeds (Table 2.2). The firing patterns used in the prescribed burns resulted in both backing and flanking fires within the plots with flame lengths up to 8 feet and 15 feet respectively (Table 2.2).

Fuels consumed in the burns were dominated by the fine litter component, with both the coarse litter and the live vegetation component contributing relatively little (Figure 2.1). Indices of fire intensity are shown in Table 2.3. For the three sites at which we had paired summer and winter burns (Orchid, Poisonwood, and Iris), all measures of fire intensity were greater for the summer burns. In the remaining five summer burns, percent scorch and char height were greater than the three winter burns. Some measures of fire intensity varied widely between the two burns done within the Locustberry and Buttonwood blocks even though they were burned on the same day. For example, the mean percent scorch was 84 in LW and 58 in LS, both burned on July 19, 2001.

Table 2.1. Dates of treatment burns and names of burns in FWS fire reports.

Site	**Treatment**	**Plot**	**Previous Burn**[1]	**Burn date**	**Burn name**
Orchid					
	Control	OC	9/11/90	---	---
	Summer	OS	9/11/90	8/16/98	AUDUBON 2A
	Winter	OW	9/11/90	12/15/98	BLVD 2
Poisonwood					
	Control	PC	7/24/86	---	---
	Summer	PS	7/24/86	8/17/98	AUDUBON 3A
	Winter	PW	7/24/86	12/15/98	HAMMOCK 2
Iris					
	Control	IC	9/10/91	---	---
	Summer	IS	9/10/91	6/22/99	BIGPINE
	Winter	IW	9/05/85	12/12/00	IRIS WINTE
Dogwood					
	Control	DC	1988[2]	---	---
	Summer	DS	1988	7/18/99	BONE2
	Winter	DW	1988	not burned	---
Locustberry					
	Control	LC	8/23/86	---	---
	Summer	LS	8/23/86	7/19/01	PALM SUMR
	Winter	LW	8/23/86	7/19/01	PALM SUMR
Buttonwood					
	Control	BC	unknown	---	---
	Summer	BS	unknown	7/18/01	BOSS SUMR
	Winter	BW	unknown	7/18/01	BOSS SUMR

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[1] Bergh and Wisby (1996)

[2] No burn shown in Bergh and Wisby (1996); Ross and Ruiz (1996) list a 1988 prescribed fire for the area (with previous burns of 1965 and 1961)

Table 2.2. Burn conditions for the 11 experimental burns. Data from FWS fire reports. ND=no data available.

Site	Code	Years since last burn	Burn Date	Days since last rain	Last rainfall amount (inches)	Wind speed (mph)	Temp. (°F)	Flame lengths (ft.)
Orchid								
	OS	8	16-Aug-98	2	0.6"	2-7	86-97	ND
	OW	8	15-Dec-98	1	0.77" (1-3 days)	3-5	68-70	1-3
Poisonwood								
	PS	12	17-Aug-98	2	0.2" 2 days and 0.4" 3 days	3-5	93	8-15
	PW	12	15-Dec-98	1	0.77" (1-3 days)	3-5[1]	68-70?	1-3
Iris								
	IS	8	18-Jul-99	3	0.71"	3-8	87	1-3, 2-6[2]
	IW	14	12-Dec-00	ND	ND	3-5	ND	1-8, 8-15
Dogwood								
	DS	11	22-Jun-99	4	2.17"	4-8	ND	1-3, 6-10
Locustberry								
	LS	14	19-Jul-01	1	0.4	ND	ND	2-6, 5-10
	LW	14	19-Jul-01	1	0.4	ND	ND	2-6, 5-10
Buttonwood								
	BS	>30	18-Jul-01	1	0.22"	ND	ND	1-4, 3-9
	BW	>30	18-Jul-01	1	0.22"	ND	ND	1-4, 3-9

[1] Assumed same as OW, because windspeeds outside of stand similar (8-16 and 9-15)

[2] First range for backing fire, second for flanking fire

Table 2.3. Indicators of fire intensity for the 11 experimental burns.

Plot	Years since last fire	Scorch (%)	Char ht. (m)	Fire temp. (°F)	Fine litter consumed (g/m^2)	Fine litter consumed (%)
OS	8	62	2.5	—	1080	71
OW	8	46	1.7	—	894	65
PS	12	66	3.0	—	1408	64
PW	12	22	1.3	—	876	52
IS	8	55	2.1	444	626	47
IW	14	39	1.3	371	—	—
DS	11	84	2.8	489	1097	74
LS	14	58	2.0	511	288	37
LW(S)	14	84	3.4	538	516	56
BS	>30	66	2.6	483	742	44
BW(S)	>30	74	2.7	523	1132	63

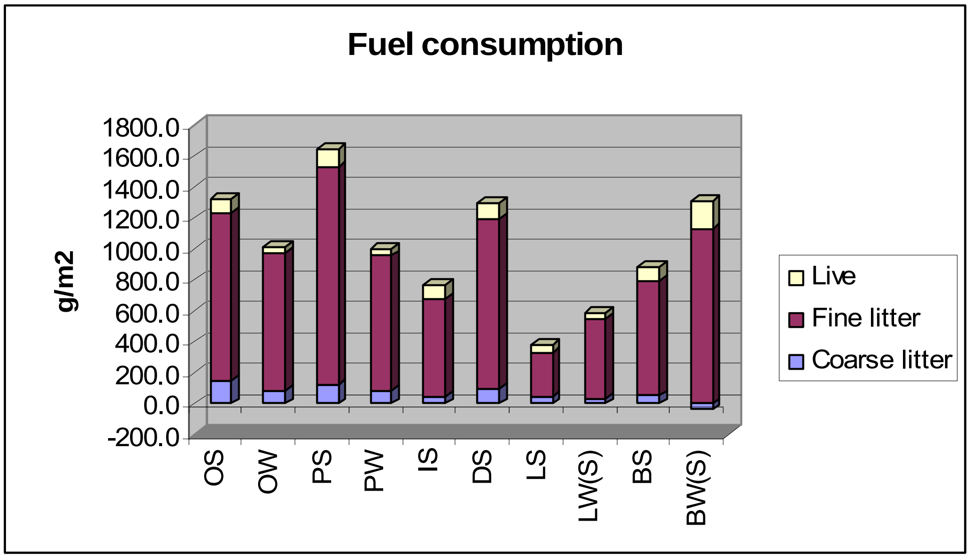


Figure 2.1. Fuels consumed in experimental burn plots. Plot codes are shown in Table 2.1. W(S) indicates that plot was originally designated for winter burn but was burned in summer.

3. TREES

Methods

In each of the 18 1.0-ha plots all trees (not including palms) with stem diameters ≥ 5 cm at 1.4 m above ground (DBH) were tagged and mapped. Trees were identified with numbered aluminum tags attached with aluminum nails at 1.4 m above ground. DBH was measured with diameter calipers. Bifurcating stems were treated separately only if there was a significant gap between them at the base (i.e. below ground attachment).

Trees were mapped by two methods. In some plots we used measuring tapes and right-angle prisms. A tape was stretched across the plot at 50-m intervals to form a baseline. A right-angle prism was used to locate trees along this axis and a second tape was used to measure the distance, up to 25 m, from the baseline to the tree. On other plots tree mapping was done with a method based on the principles of triangulation, and adhering to the guidelines of Boose et al. (1998). In this method, a target tree with unknown coordinates is located, its tag number recorded, and a series of 3 consecutive clockwise distance measurements are taken from reference benchmarks (0,0; 0,8; 6,0 m along plot boundaries). Once at least three targets have been mapped, benchmarks can be abandoned and the known trees are then used as reference trees for other unknown targets. The angles from references to targets cannot exceed 120° or be below 20°. Trees downed by Hurricane Georges were mapped from their original base, their direction of fall recorded, and length measured. Data were analyzed using INTERPOINT software (Boose et al. 1998).

Postburn canopy scorch, bole char height, and scarring measurements were taken within one month following the fires. Percent scorch is an estimate of the amount of live needles killed by the fire using the following scale: 0%, 10%, 25%, 50%, 75%, 90%, 100% (percent crown scorch volume, Fowler and Sieg 2004). There was essentially no needle consumption in any of the fires. Maximum char height was directly measured on individual trunks, with directional orientation noted (8 possible compass directions: N, NE, E, SE, S, SW, W, NW). The height of basal scarring ("cat faces") was also noted, along with its direction. Trees in burned plots and their associated control plots were checked for mortality annually.

In the plots in which the spatial distribution of all trees was known, the spatial pattern of fire intensity, as reflected in individual tree crown scorch and bark char height, could be determined by interpolation of the point measurements. We interpolated the data via a kriging algorithm (Maclean and Cleland 2003; Taylor et al. 2004). Using the Geostatistical function in ArcView 3.1, ordinary kriging was performed in two steps: semivariogram analysis (Isaaks and Srivastava 1989) and gridding. A semivariogram is a function fit to variances associated with all pairs of point measurements within bracketed zones across the range of distances present in the data. We applied a spherical model to fit semivariograms for both scorch percentage and char height. A search distance of 15m was used in the gridding process. The output was a map of 1 m x 1m grid cells with an estimated scorch percentage and char height in each cell.

Results and Discussion

Initial Conditions

Hardwood species generally make up a minor part of the pine rockland tree layer. There were few hardwood trees in the open sites, and none in the Orchid control plot. Hardwoods were most numerous at the Buttonwood site and in the Buttonwood control plot the number of hardwood stems approached the number of pine trees (Figure 3.1).

Metopium toxiferum, poisonwood, is the dominant hardwood species in the Keys pinelands, as it constitutes 75.6% of all hardwood stems. It is found in every plot except the Orchid control plot which is devoid of hardwood trees (Table 3.1). Other, rather minor, species include *Conocarpus erectus*, *Myrica cerifera*, *Piscidia piscipula*, and *Pithecellobium guadalupense*. A total of 18 hardwood tree species were found in the plots, and all but one species occurred in the Buttonwood control plot. The Buttonwood site had both the greatest number of hardwood trees and the highest tree species diversity. The three shrubby sites (Poisonwood, Dogwood, and Buttonwood) averaged 122±41 hardwood trees/ha and 6.7±1.6 species/ha while the open sites (Orchid, Iris, Locustberry) averaged 8±2.9 trees/ha and 1.9±0.5 species/ha.

The size-class distribution of both pines and hardwood species are shown in Figure 3.2. Hardwoods were generally confined to the two smallest size classes (≤15 cm DBH). The largest diameter *Metopium* was 22.8 cm in the Poisonwood control plot.

The density of pine trees ranged from 361 ha^{-1} in the Iris summer plot to 1006 ha^{-1} in the Dogwood winter plot (Figure 3.1); the mean pine density was 570 ± 37.4 trees ha^{-1}. The high density of pines in the Dogwood winter plot was due to very large number of pines in the smallest size class (Figure 3.2). The largest size-classes were best represented in the Poisonwood plots. The largest pines in the study were trees with DBH of 40.4 cm (PW), 39.0 cm (PW), 38.6 cm (PC), and 38.0 cm (PS).

Fire effects

We were able to evaluate postburn mortality of trees the year following the first seven burns (Table 3.2). The aboveground portions of hardwood trees were frequently killed by the experimental fires, although in many cases the topkilled individual survived and resprouted from belowground parts. In the sites with significant numbers of hardwoods, the mortality, or topkilling, of trees ranged from 21% after the Poisonwood summer burn to 97% after the Dogwood summer burn.

Pine mortality was higher after the summer burn than the winter burn in the three sets of paired burns (Table 3.2). At the Orchid site the difference was slight, but at Poisonwood and Iris, the mortality after summer burns was >3 times the mortality after winter burns. The Dogwood summer burn was the most severe burn in terms of pine mortality, with 57.1% of the trees dying.

Mortality of pines was related to diameter at the Orchid and Dogwood sites, with smaller trees more likely to die; there was no statistical relation between size and mortality at the Poisonwood and Iris sites (Table 3.3). In all seven of the burns, the degree of scorching of the tree crown was highly related to tree mortality whereas the height of bark char was significant only in four of the seven burns (Table 3.3).

The spatial pattern of pine mortality related to crown scorch and char height is shown for the Orchid summer and winter burns in Figure 3.3. The pattern of pine mortality appeared to track local variation in fire intensity quite closely, though not uniformly for both indices of intensity. For example, the high mortality of smaller trees in the northwest corner of the Orchid winter burn plot appeared to be more highly related to severe crown scorch than to the height of bark charring.

Table 3.1. Hardwood tree species diversity and number of stems ≥5 cm dbh per ha in the study plots.

	Orchid			Poisonwood			Iris			Dogwood			Locustberry			Buttonwood		
	OC	OS	OW	PC	PS	PW	IC	IS	IW	DC	DS	DW	LC	LS	LW	BC	BS	BW
No. of species	0	2	1	3	5	4	4	2	1	6	2	4	4	1	2	17	11	8
No. of stems	0	8	6	26	54	74	28	5	8	65	62	60	15	2	3	419	204	135
Species																		
Metopium toxiferum		6	6	24	50	50	22	3	8	51	61	54	10	2	2	283	159	96
Conocarpus erectus		2			1	18	1									53	3	1
Myrica cerifera				1	1	4		2		8						18	10	10
Psicidia piscipula				1	1		4						3		1	10	13	20
Pithecellobium guadalupense					1					2						3	10	4
Byrsonima lucida						2				1	1	3				4	4	
Coccoloba uvifera																13	1	
Guapira discolor																10	1	2
Rhizophora mangle																10		
Manikara bahamensis																6	1	
Guettarda scabra												2				1	1	1
Myrsine floridana										2			1			1		
Psidium longipes							1			1						1	1	
Reynosia septentrionalis																3		
Bumelia salicifolia												1	1					
Pisonia rotundata																1		1
Bumelia celastrina																1		
Ficus citrifolia																1		

Table 3.2. Percent mortality of trees after experimental burns. In many cases, hardwood species resprouted from the base of the tree.

Site	Burn	Pine	Hardwoods
Orchid	Summer	17.6	37.5*
	Winter	16.0	50.0*
Poisonwood	Summer	28.4	21.2
	Winter	7.8	29.4
Iris	Summer	5.6	60.0*
	Winter	1.8	0*
Dogwood	Summer	57.1	96.8

* <10 stems preburn

Table 3.3. Mean DBH, scorch, and char height of surviving pine trees and trees killed by burns in seven of the experimental plots, with ANOVA p-value.

	DBH (cm)			Scorch %			Char height (m)		
	Live	Dead	p-value	Live	Dead	p-value	Live	Dead	p-value
OS	13.00	10.53	<0.001	55.1	90.1	<0.001	2.33	3.22	<0.001
OW	11.93	9.49	<0.001	37.4	91.6	<0.001	1.56	2.24	<0.001
PS	13.45	12.66	0.238	55.5	93.1	<0.001	2.57	3.77	<0.001
PW	13.43	14.96	0.289	19.1	52.7	<0.001	1.29	1.48	0.361
IS	15.17	13.42	0.229	53.2	85.0	<0.001	2.05	2.50	0.164
IW	11.18	10.87	0.847	38.4	86.5	<0.001	1.30	1.55	0.421
DS	10.95	9.80	0.002	64.2	99.1	<0.001	1.81	3.60	<0.001

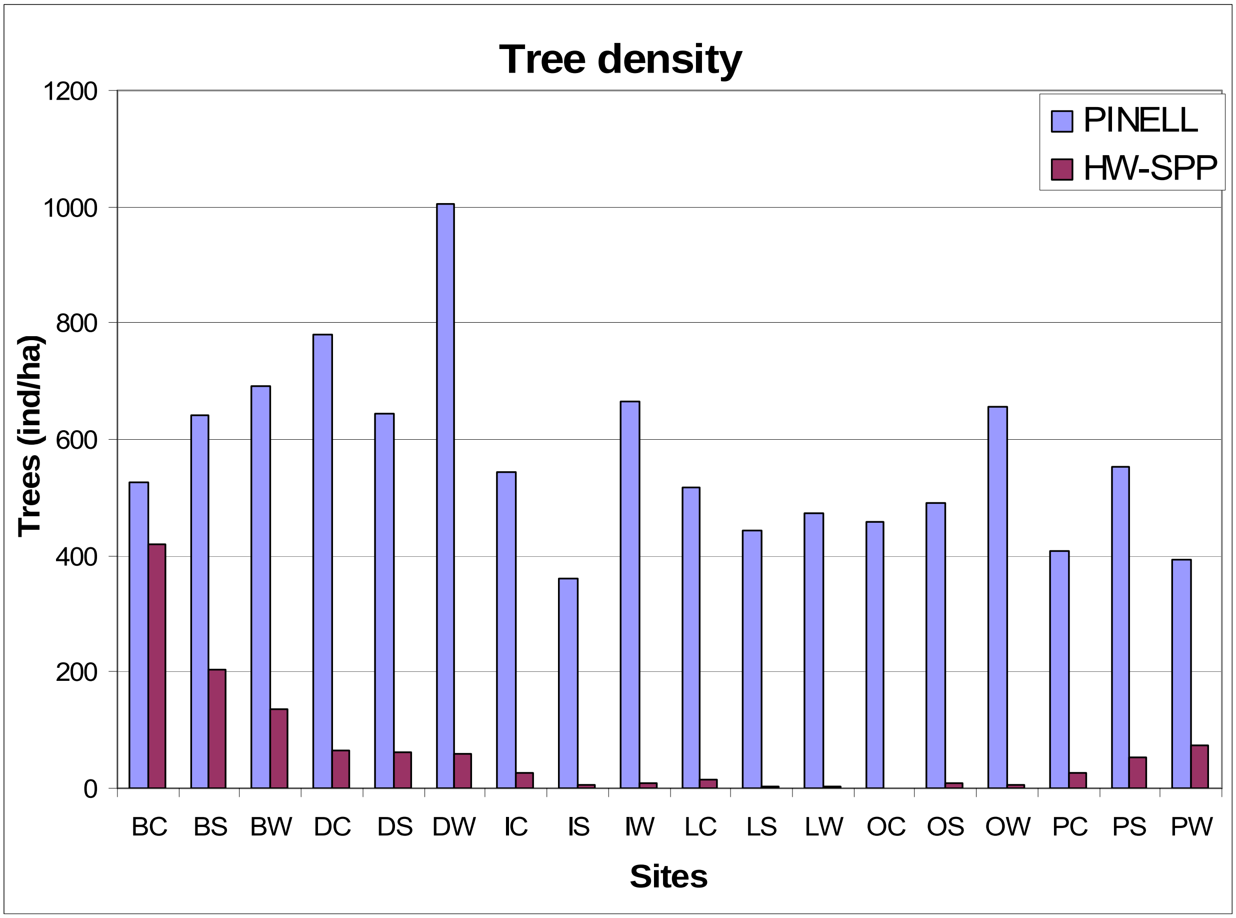
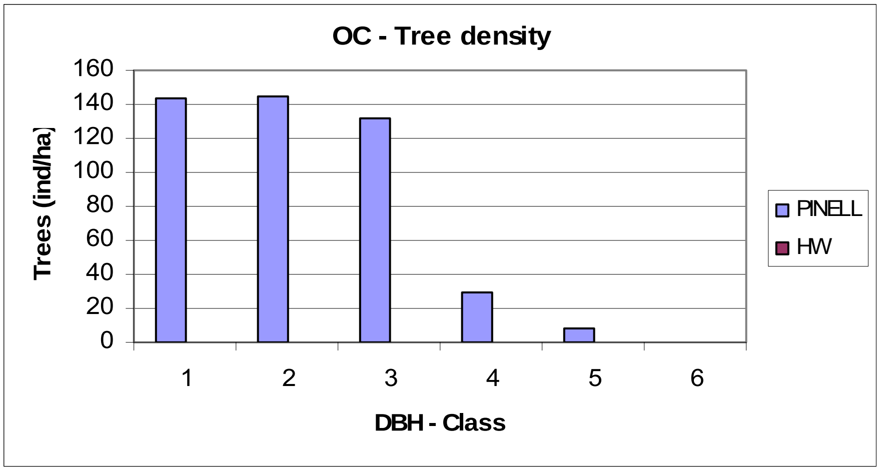
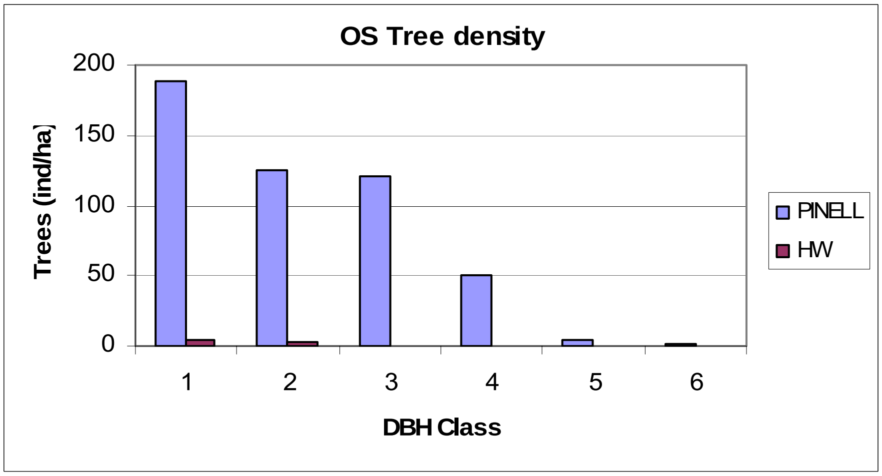


Figure 3.1. Density of pines and hardwood species in the 18 experimental plots.





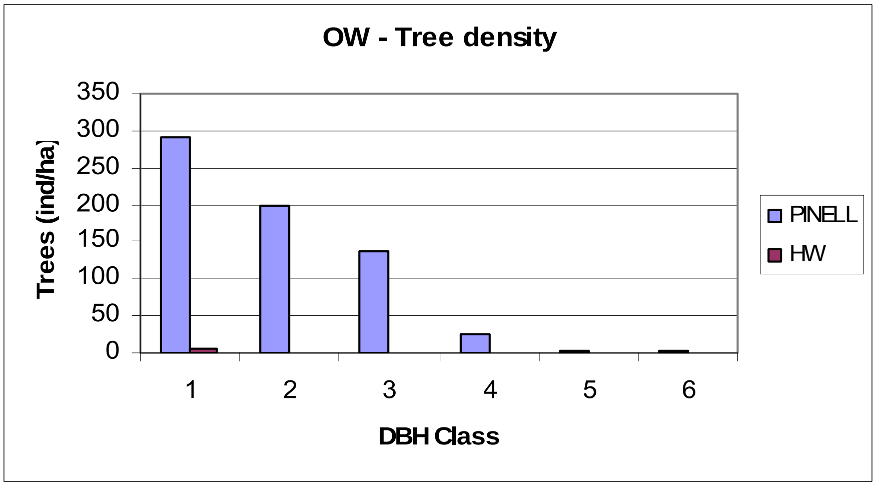
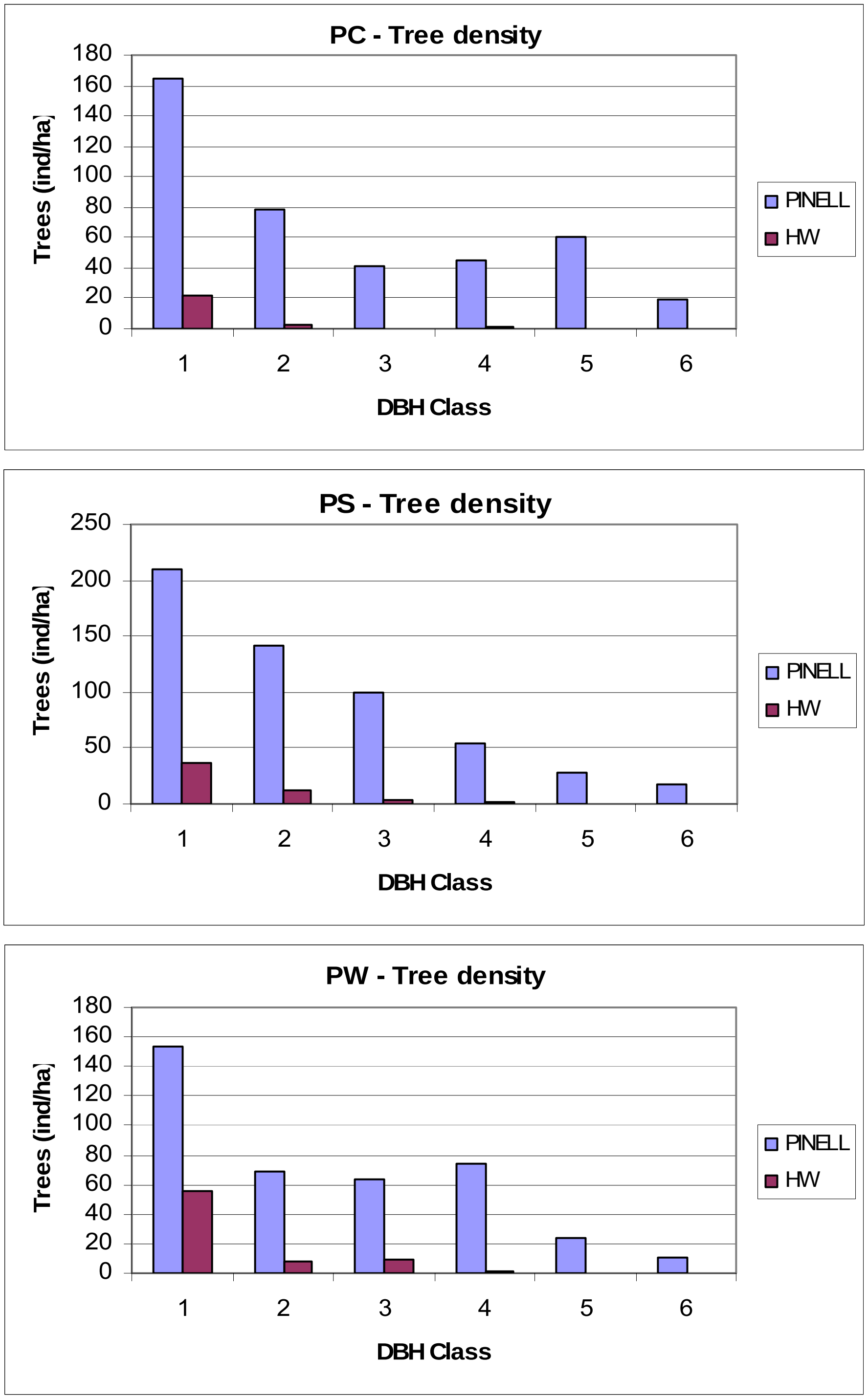


Figure 3.2. Size-class distributions of pines and hardwoods in the study plots. DBH class 1 = 5-10 cm, 2 = 10.1-15.0 cm, 3 = 15.1-20.0 cm, 4 = 20.1-25.0 cm, 5 = 25.1-30 cm, 6 = > 30 cm.



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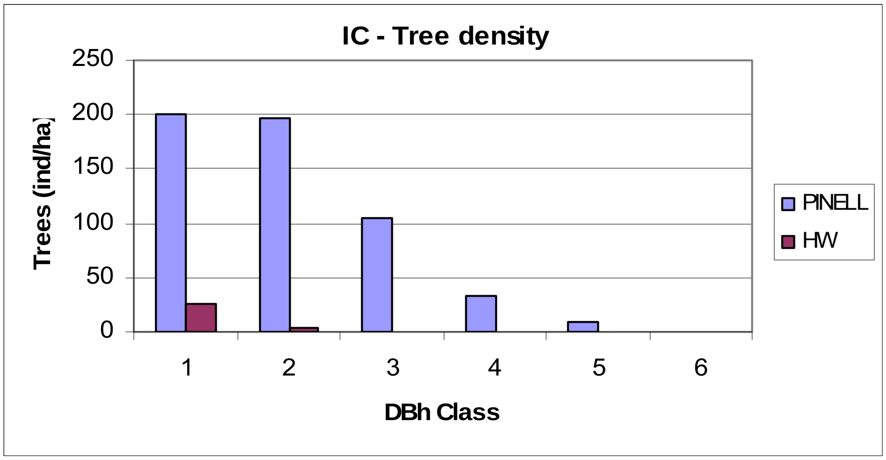

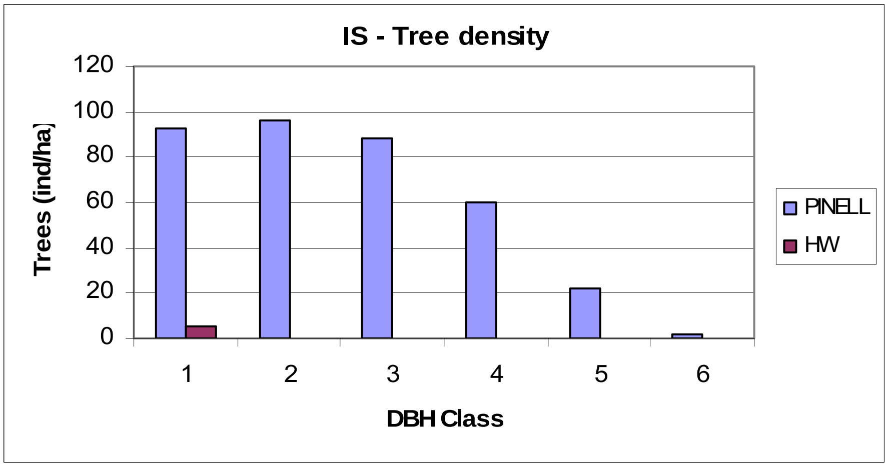

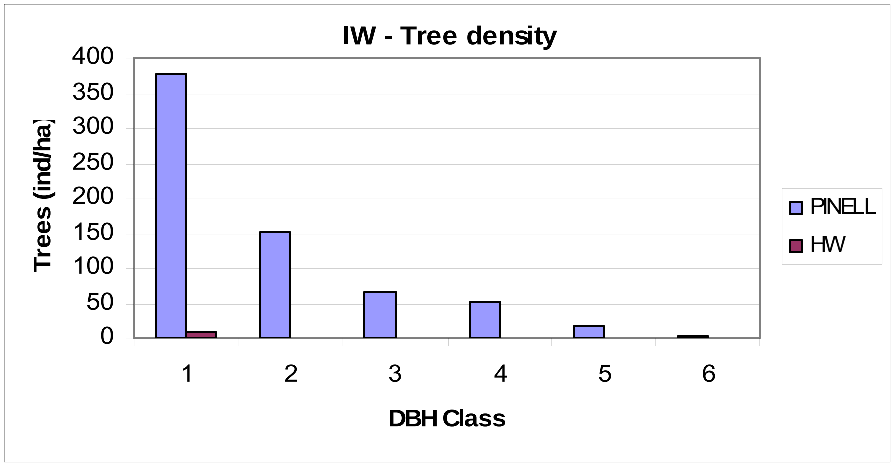

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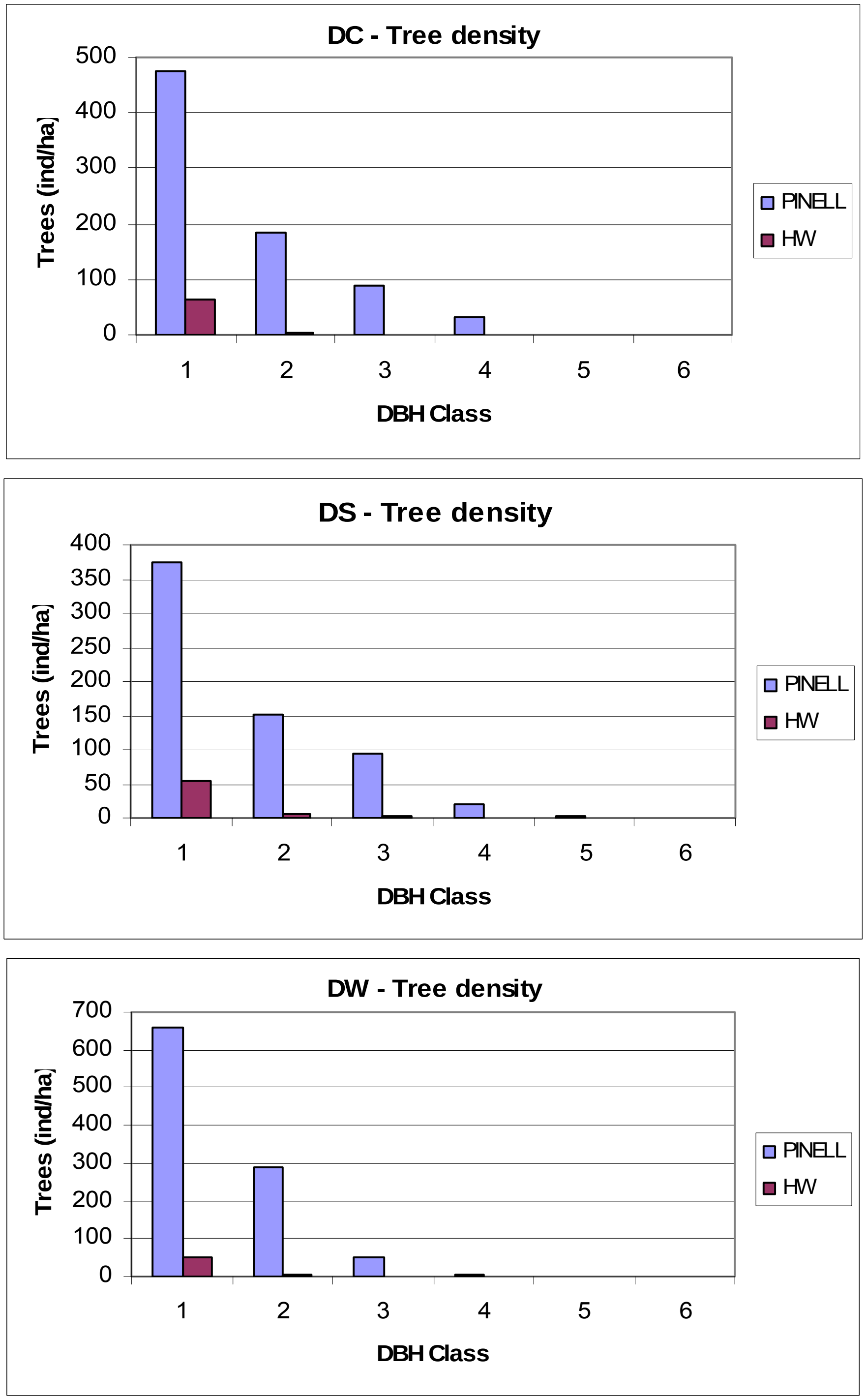

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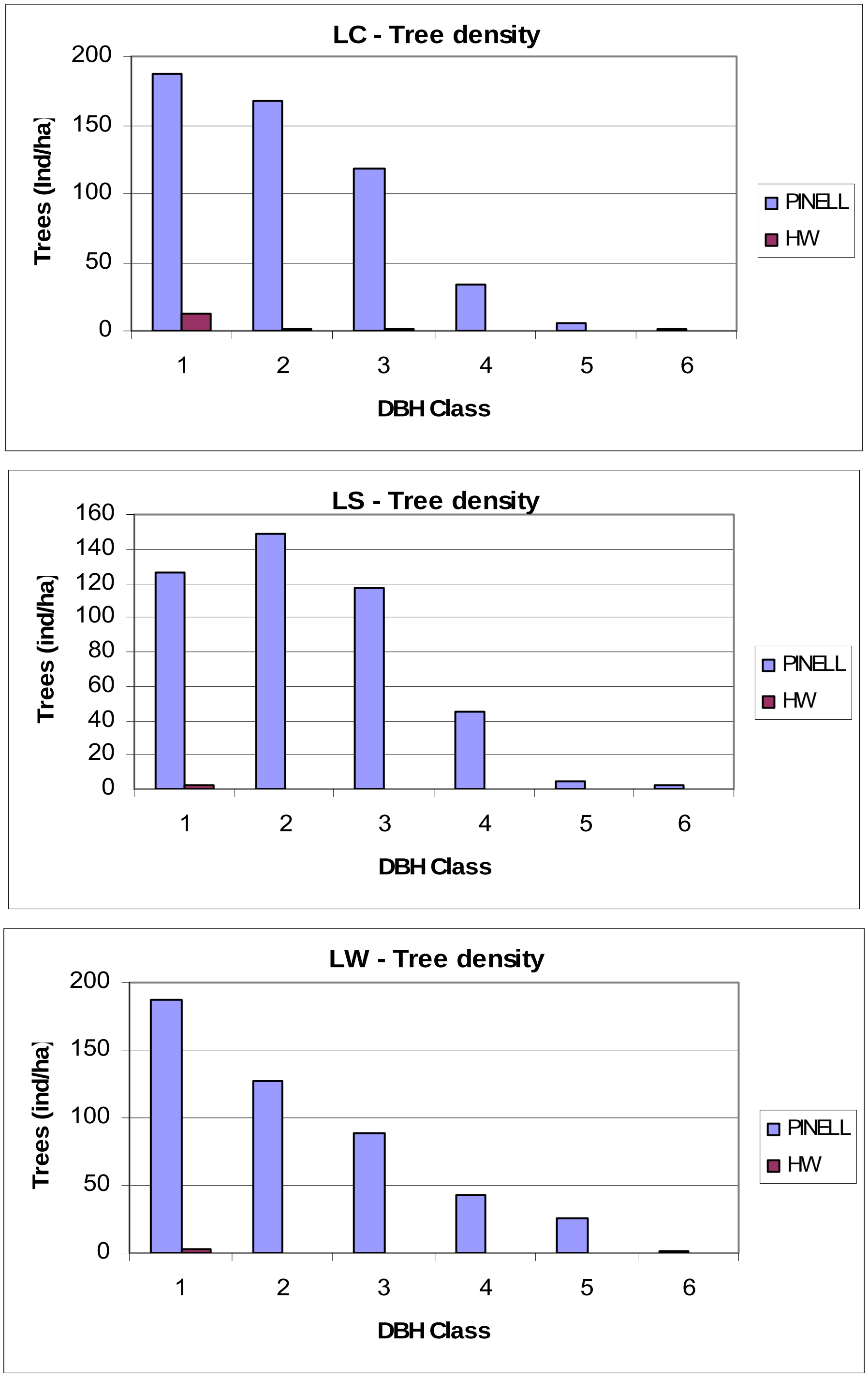

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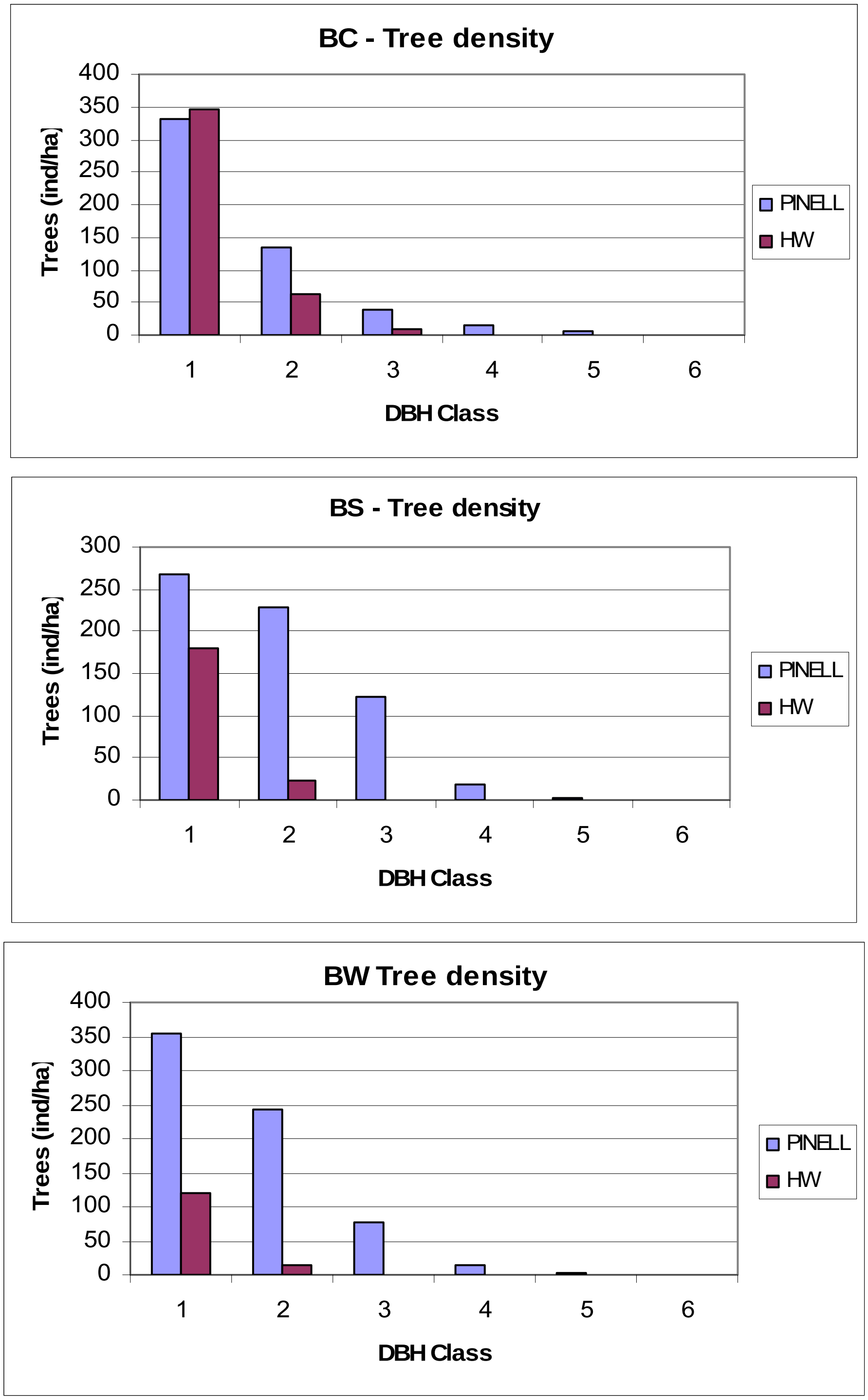

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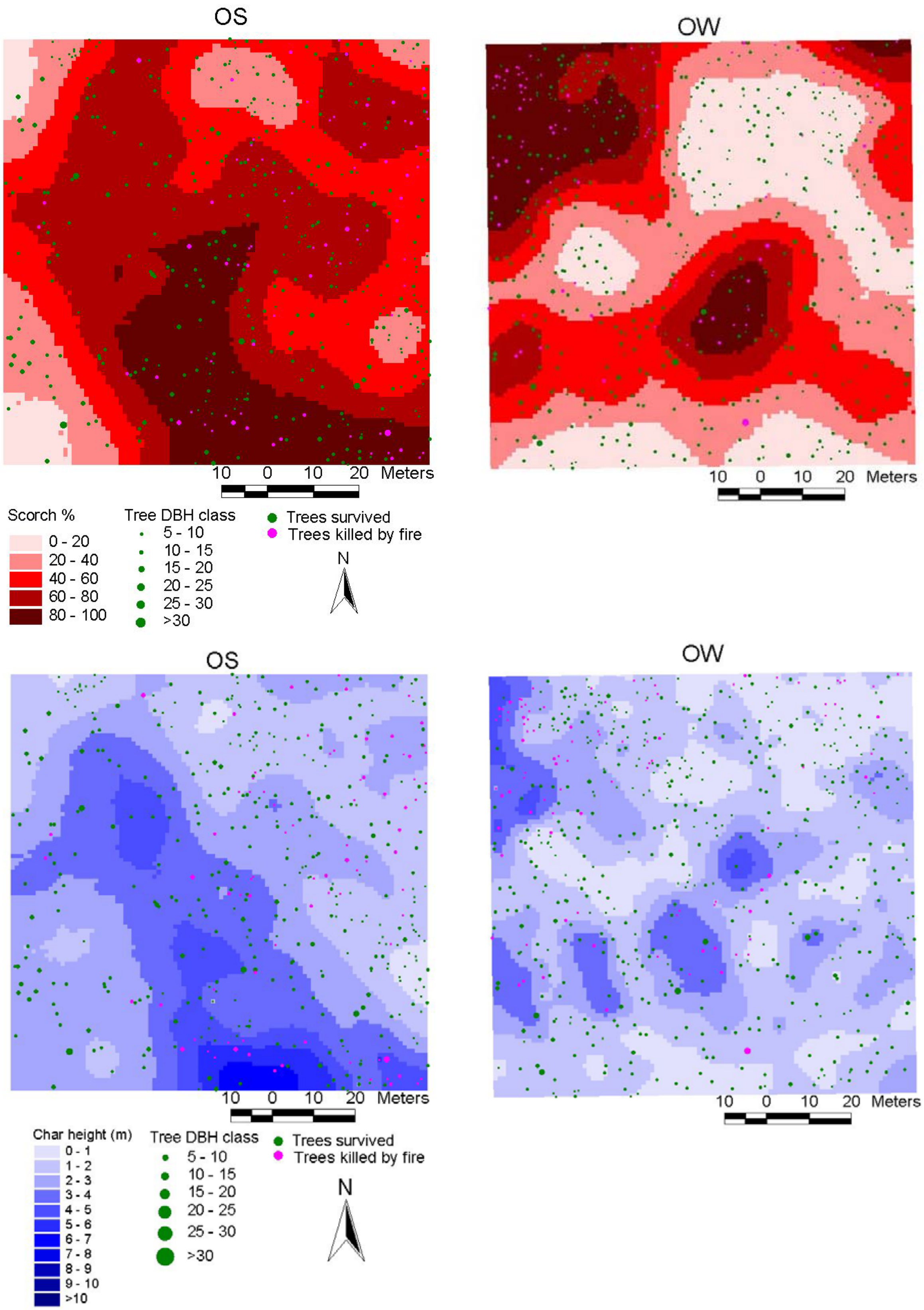


Figure 3.3. Maps of canopy scorch and char height in Orchid winter and summer burn plots. See text for methods. Locations and size classes of live and dead pine trees shown.

4. PALMS

Methods

Palms were not sampled in the same manner as trees. All palms with a total height (height of the tallest leaf) ≥ 1 m were measured within the 20 shrub plots within each 1.0-ha plot before burning. For each individual we recorded the species, the total height, and the height of the apical meristem, or stem height. Palms were re-sampled at one-year intervals after the burns on the same schedule as hardwood shrubs: Orchid and Poisonwood plots were sampled four times, the Iris and Dogwood control and summer burn plots were sampled three times, the Iris winter burn plot was sampled twice and the Dogwood winter and all the Locustberry and Buttonwood plots were sampled only once before the burns.

Our method of recording palms before and after fire allowed us to determine fire-caused mortality of only those palms whose apical height was at least 1 m. For palms with apical heights < 1 m, it is possible that after the burn reduced leaf size, or drooping leaves, might cause a plant to not be counted even though the plant was still alive. Height growth of *Coccothrinax* and *Thrinax* is rather slow (Cooley 2004), so that by comparing the density of palms with stems ≥ 1 m tall before the burn to the density 1 yr after the burn, we can estimate the number of palms killed by the fire. The growth of palms into the ≥ 1m size class during the year after the fire could contribute to a minor underestimation of mortality.

Results and Discussion

Initial Conditions

Thrinax morrissii, the Keys thatch palm, and *Coccothrinax argentata*, the silver palm, both tropical species, are the dominant palms in the Keys pinelands and are common in all of the plots (Table 4.1). *Serenoa repens*, saw palmetto, is a temperate palm with an unusual horizontal creeping stem. It is a codominant in the Iris and Poisonwood sites but is present in all the study plots. A fourth species, also temperate in distribution, *Sabal palmetto*, the cabbage palm, is infrequent on Big Pine Key.

The density of palms ≥1m in height ranged from <2000 to >5000 individuals per hectare. Densities were generally <3000/ha in open sites and >3000/ha in shrubby sites. The highest density was in the Poisonwood control plot, with 5150 palms/ha and the lowest was in the Orchid control plot, with 1880 palms/ha. Overall, the mean *Thrinax* density was 1600 palms/ha and the density of *Coccothrinax* was 1000 palms/ha. *Thrinax* densities were greater than *Coccothrinax* in all plots except the Poisonwood control and the Iris winter plots; *Thrinax* was six times more abundant than *Coccothrinax* in the Orchid site. *Serenoa* reached its highest densities, 1760 palms/ha, at the Iris site. In the lower Keys pinelands *Serenoa* is generally found around the edges of depressions or the wetter parts of pinelands. *Sabal* is found sporadically in the plots and only 21 individuals were encountered in 360 shrub plots.

The size–class distributions of *Thrinax* and *Coccothrinax* are shown in Figure 4.1. Typically there is a decrease in the number of individuals as the size increases. Exceptions include *Thrinax* in PC, BC, and BW plots and *Coccothrinax* in IC, LC, and BW plots. *Thrinax* dominates the larger size classes. The tallest *Thrinax* recorded in the study was 5.50 m tall and the tallest *Coccothrinax* was 4.05 m. The leaves and stem diameter of *Thrinax* are also larger so it has a larger mass than *Coccothrinax* (Cooley 2004).

Fire Effects

Mortality of *Thrinax* and *Coccothrinax* is shown in Figure 4.2 for the seven burns in which one-year postburn data were collected. There were three pairs of summer/winter burns and in two of the cases there was higher mortality due to the summer burns (Orchid and Poisonwood) and in one case there was higher mortality after the winter burn (Iris). The seasonal pattern was the same for both species of palm, although the mortality of *Coccothrinax* was substantially greater than *Thrinax* at the Orchid site. The most striking difference between the two species was in the Dogwood summer burn in which more than half the *Coccothrinax* died, but only 2.7% of the *Thrinax* were killed.

The weighted mortality (summing the palms killed in the plots and dividing by the number of pre-burn palms) is shown in Table 4.2. There was little difference between summer and winter burns for *Thrinax*, but there was substantially greater mortality after summer burns for *Coccothrinax*. Overall mortality of *Coccothrinax* was about three times greater than that of *Thrinax* (32.5% vs. 10.3%).

Cooley's (2004) study provides partially independent estimates of mortality because they are based on marked individuals in the four plots burned the last year of the project (Buttonwood and Locustberry plots) as well as the Iris winter plot which is included in our analysis. Based on plants with apical heights > 1m, Cooley found 20.9% mortality of *Coccothrinax* and 9.6% mortality of *Thrinax*, somewhat less than in the results reported here. The high mortality rates for these tropical palms differ from general lack of mortality seen when *Serenoa* (Abrahamson 1984) and *Sabal* (McPherson and Williams 1998) are burned.

Because our sampling protocol for palms was based on individuals with leaves at least 1 m above ground level we cannot determine whether reductions in numbers of palms in the smaller size classes was due to mortality or simply reduced leaf size after burns. There is mortality of these smaller individuals based on field observations and Cooley's (2004) results based on marked plants. Cooley found lower mortality for plants with stem heights ≤ 1 m than for larger plants. Based on 121 marked *Thrinax* and 144 marked *Coccothrinax*, mostly from the Locustberry and Poisonwood plots burned in the summer, Cooley documented 11.1% mortality for *Coccothrinax* and 6.6% mortality for *Thrinax*.

While it is not possible to determine mortality of smaller individuals by our sampling method, we are able to compare the impact of burning on the numbers of individuals reaching certain size limits. The postburn recovery of smaller *Thrinax* and *Coccothrinax*, those which met our sampling criterion of ≥1 m total height and whose stem height is ≤ 1m, is shown in Figure 4.3. In the three pairs of summer and winter burns, the recovery of palms one year after burning was always greater after the summer burn, this being most pronounced at the Iris site. The same pattern held after three years at the Orchid and Poisonwood sites. This result indicates that there are different factors affecting larger palms than smaller palms because in the Orchid and Poisonwood plots there was greater mortality of plants with stems > 1 m tall after summer burns than winter burns.

Serenoa has the unique habit among North American palms of a horizontal, branching stem. Because of this habit, most of the stems are less than 50 cm above ground, and in this study only 2 individuals had apical meristems > 1 m above ground level. Therefore the postburn recovery of *Serenoa* can be compared to that of the smaller *Thrinax* and *Coccothrinax*. Unlike the two common tropical palms, there is greater recovery of *Serenoa* one year after winter burns at the Poisonwood and Iris sites (Figure 4.4). To add additional confusion, however, the pattern is reversed by the third year at the Poisonwood site so that the summer burn has greater recovery. Therefore, at the two sites where we have paired summer and winter burns and three years of postburn data, the smaller palms recover more after summer than winter burns by year three (Figures 4.3 & 4.4).

Serenoa appears to be more fire-adapted than *Thrinax* or *Coccothrinax*, based on mean percent recovery of stems < 1 m tall one year after burning: *Serenoa* 86.0±4.27, *Thrinax*, 64.3±4.94; and *Coccothrinax*, 55.6±4.77.

Table 4.1. Density of palms (total height ≥ 1 m) in the study plots.

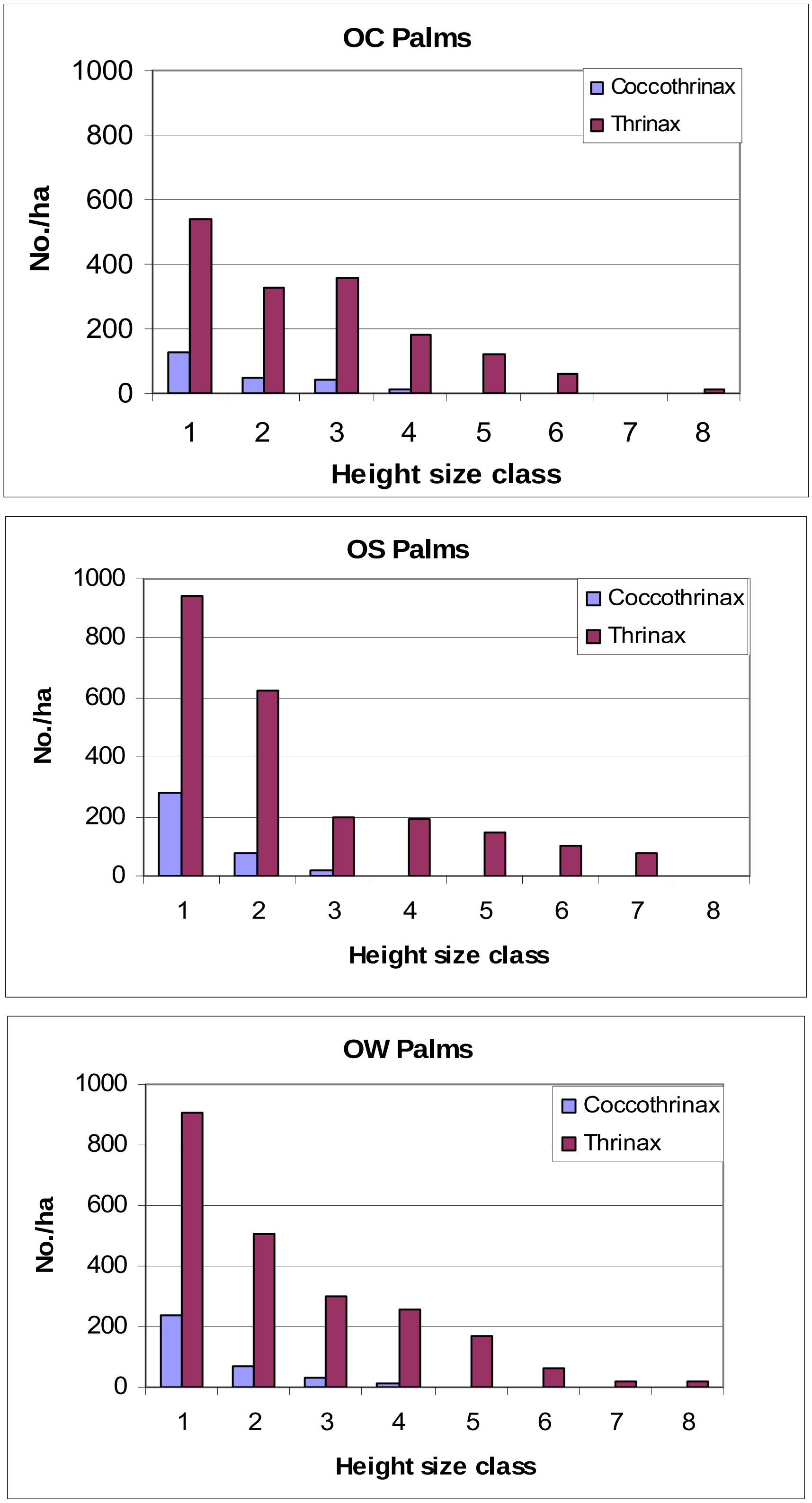
	Orchid			Poisonwood			Iris			Dogwood			Locustberry			Buttonwood			
	OC	OS	OW	PC	PS	PW	IC	IS	IW	DC	DS	DW	LC	LS	LW	BC	BS	BW	Mean±SE
Thrinax morrisii	160	231	225	185	159	126	137	68	62	199	148	186	151	178	164	257	208	204	169±12.1
Coccothrinax argentata	23	38	35	300	158	132	32	31	79	106	105	104	63	64	92	101	196	134	99±16.3
Serenoa repens	5	58	8	30	97	106	83	176	128	10	29	4	6	11	7	2	17	18	44±12.2
Sabal palmetto	0	0	2	1	1	1	2	4	10	0	0	0	0	0	0	0	0	0	1.2±0.58
Total	188	327	270	516	415	365	254	279	279	315	282	294	220	253	263	360	421	356	

Table 4.2. Mortality (%) of palms with stems ≥ 1 m tall.

	Summer burns	Winter burns	All burns
Thrinax	10.8	9.3	10.3
Coccothrinax	41.0	15.0	32.5

Figure 4.1. (Following six pages). Size-class distribution of *Thrinax* and *Coccothrinax* in the study plots. Size-class based on total height, or height of tallest leaf.

Size-classes:
1=1.0-1.49m
2=1.5-1.99m
3=2.0-2.49m
4=2.5-2.99m
5=3.0-3.49m
6=3.5-3.99m
7=4.0-4.49m
8=4.5-4.99m



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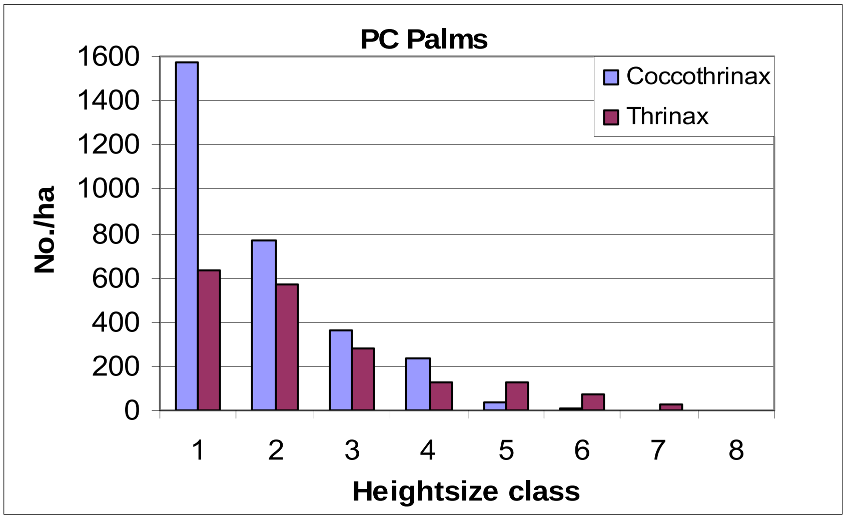

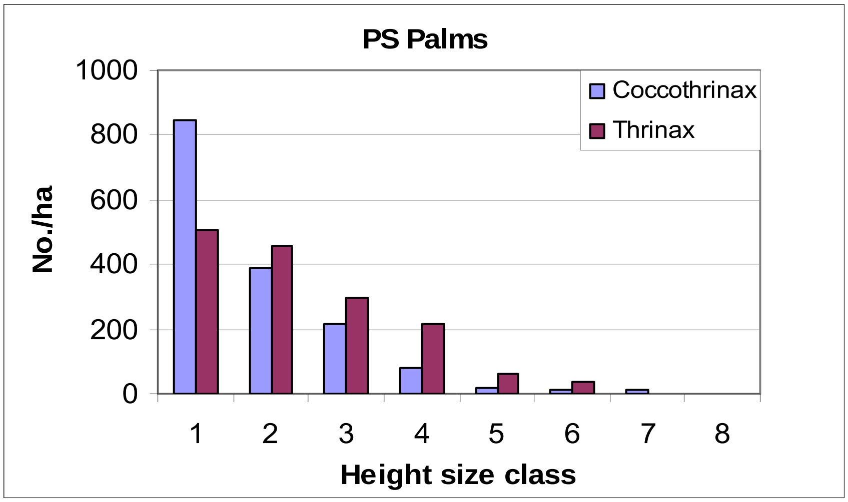

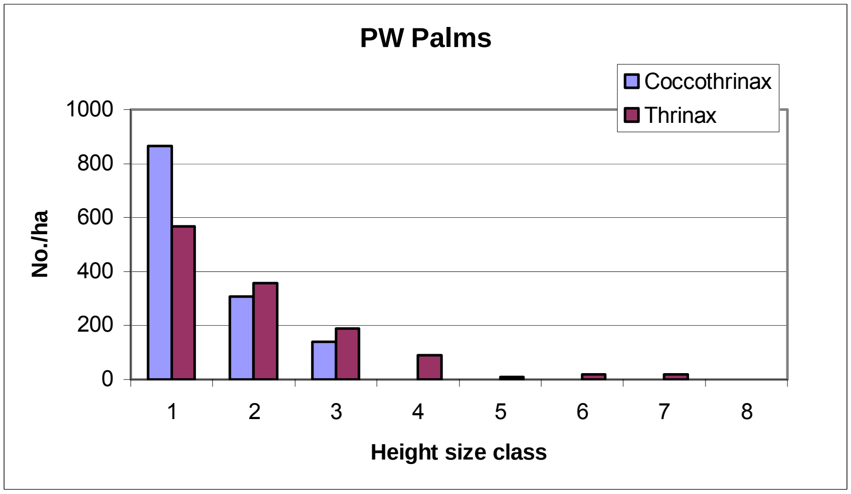

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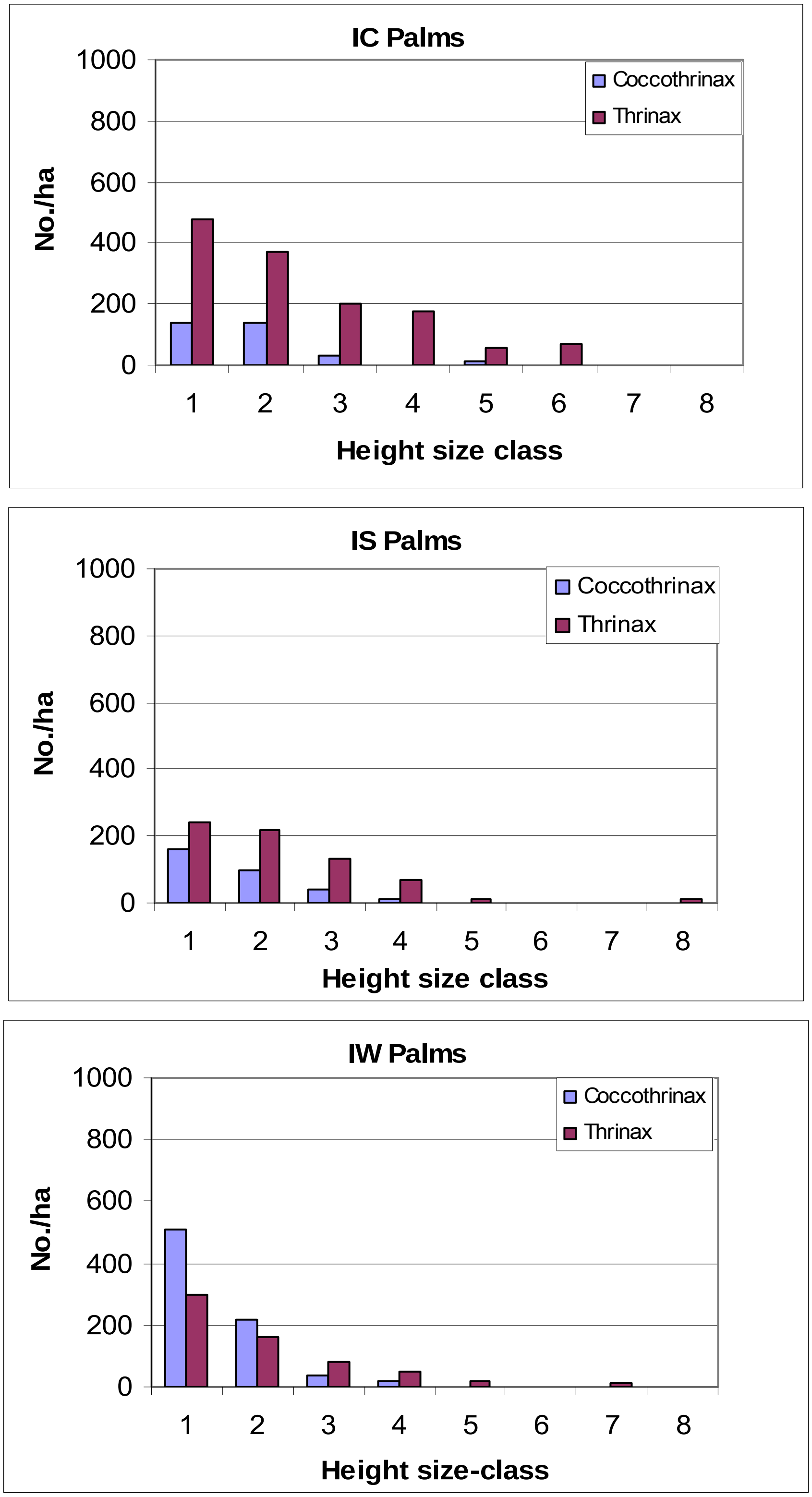

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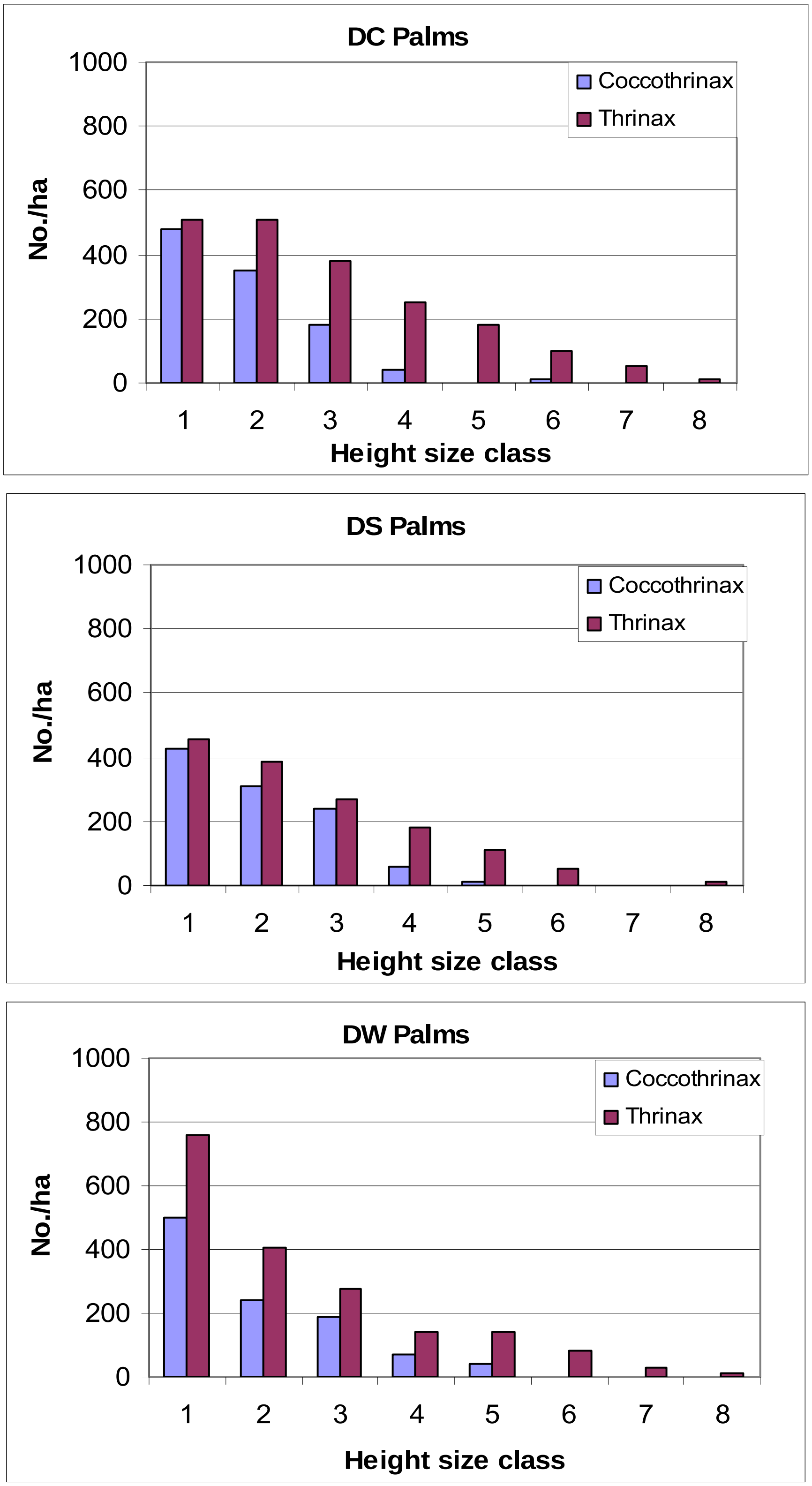

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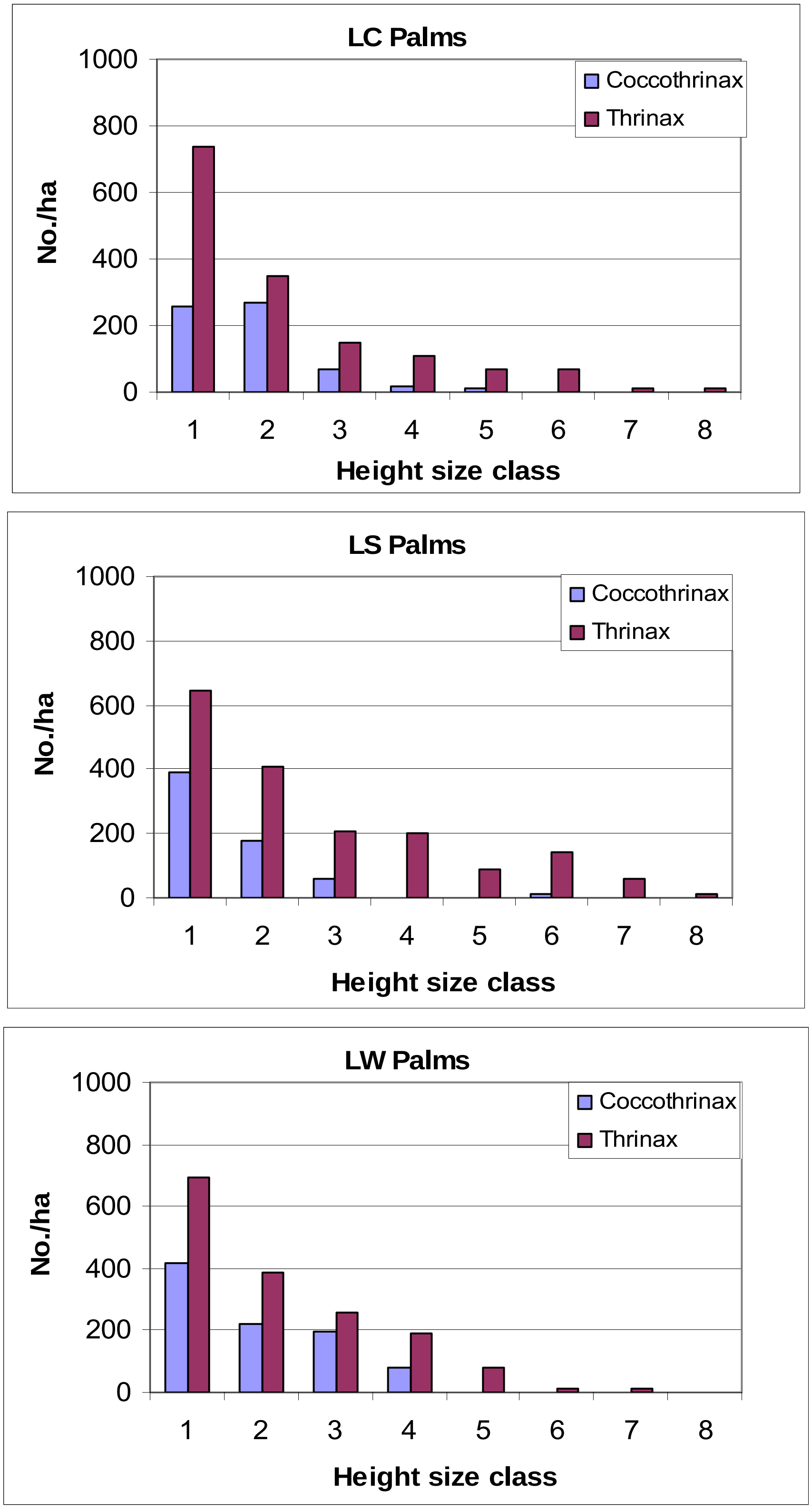

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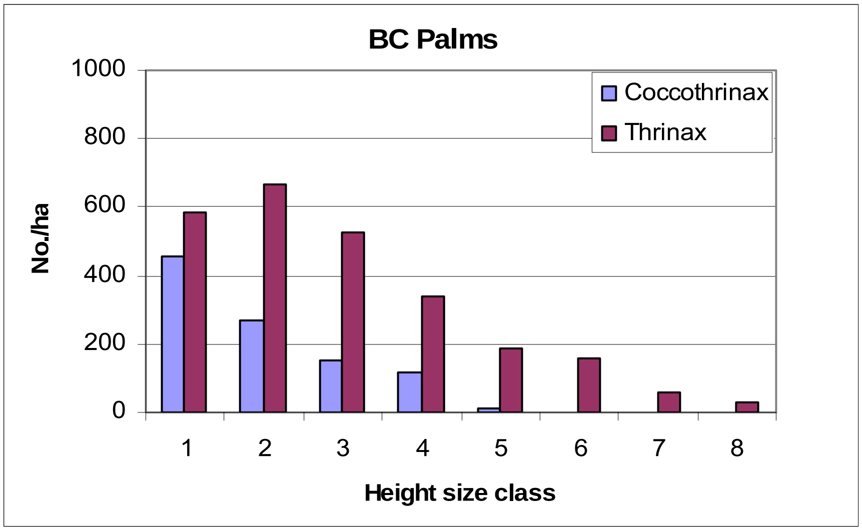

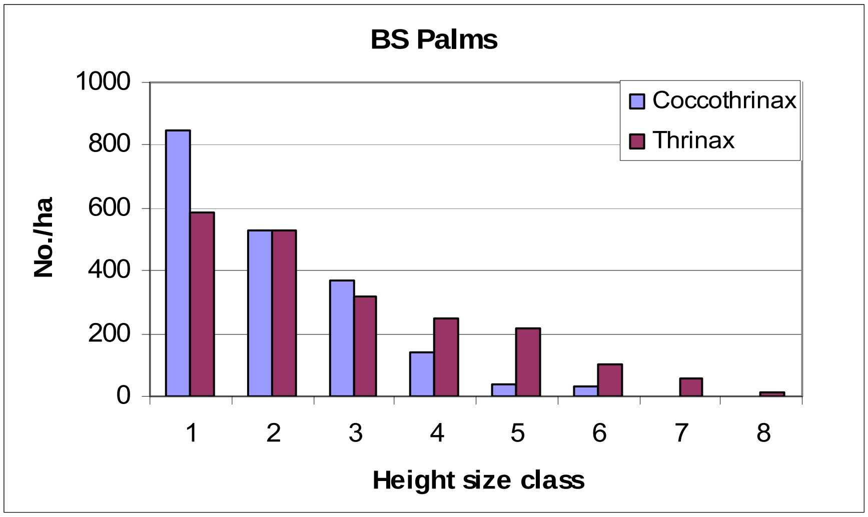

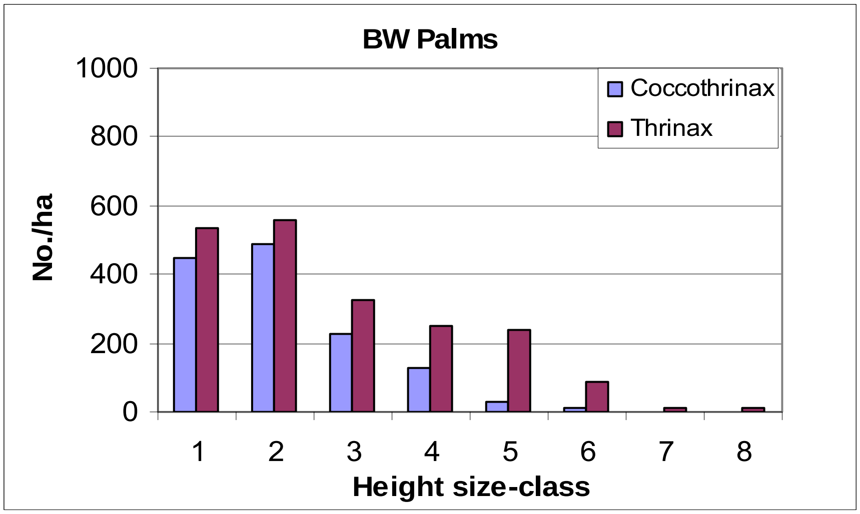

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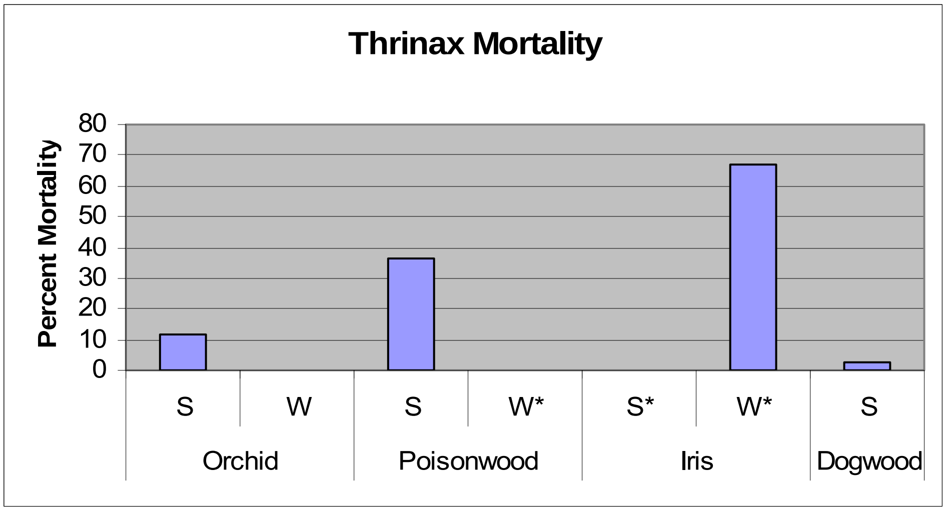


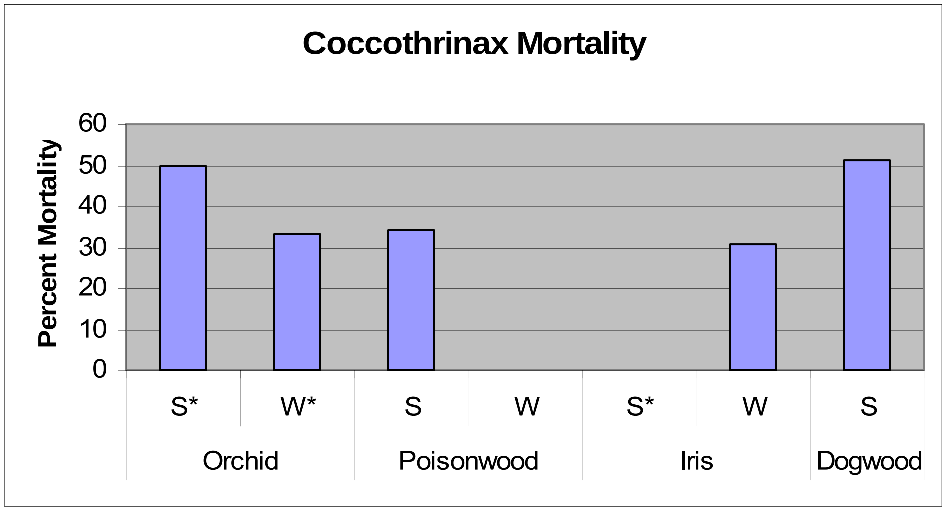
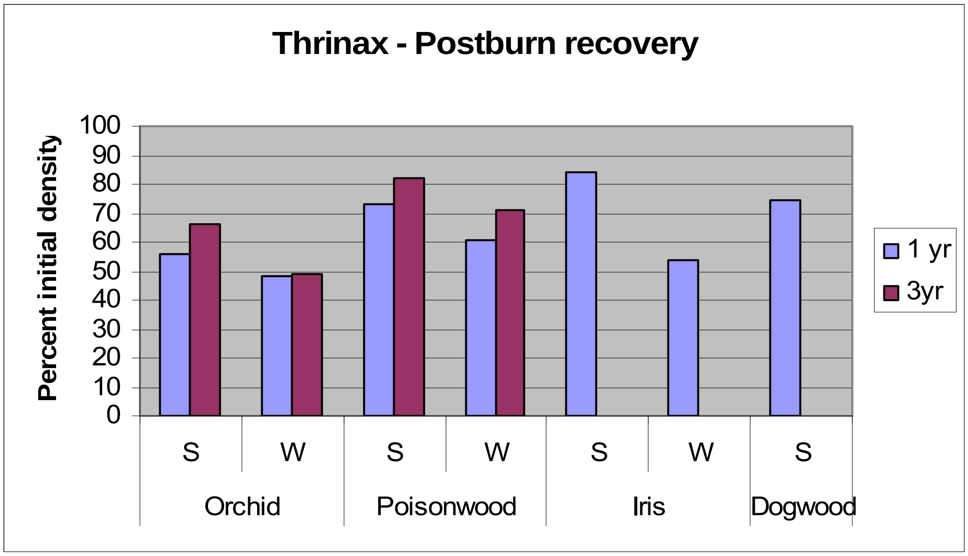


Figure 4.2. Mortality of palms with apical heights ≥ 1 m. *Plots with < 10 individuals.



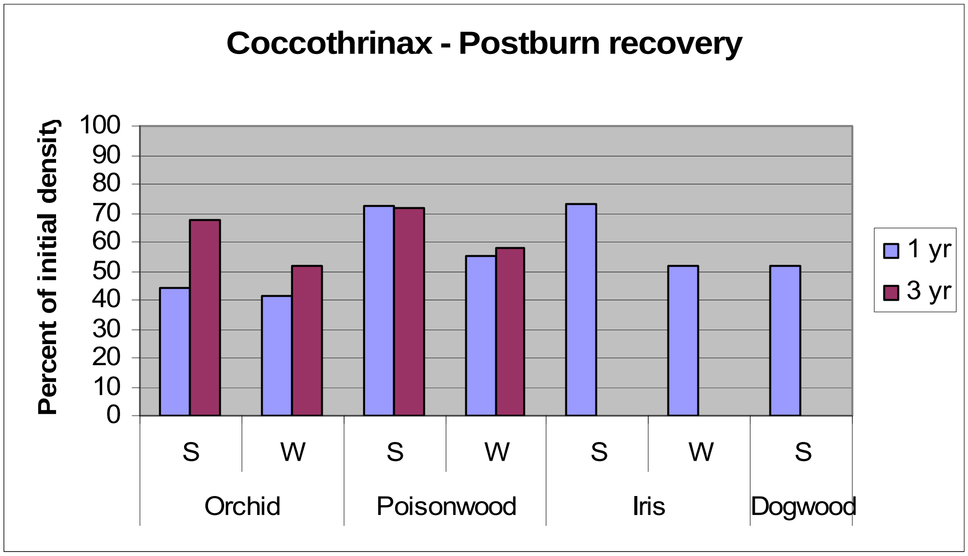


Figure 4.3. Number of palms with total heights ≥ 1 m and apical heights ≤ 1 m one and three years after burns, expressed as percentage of number of palms meeting these criteria before the burns. The Iris and Dogwood plots were not sampled 3 years after burning.

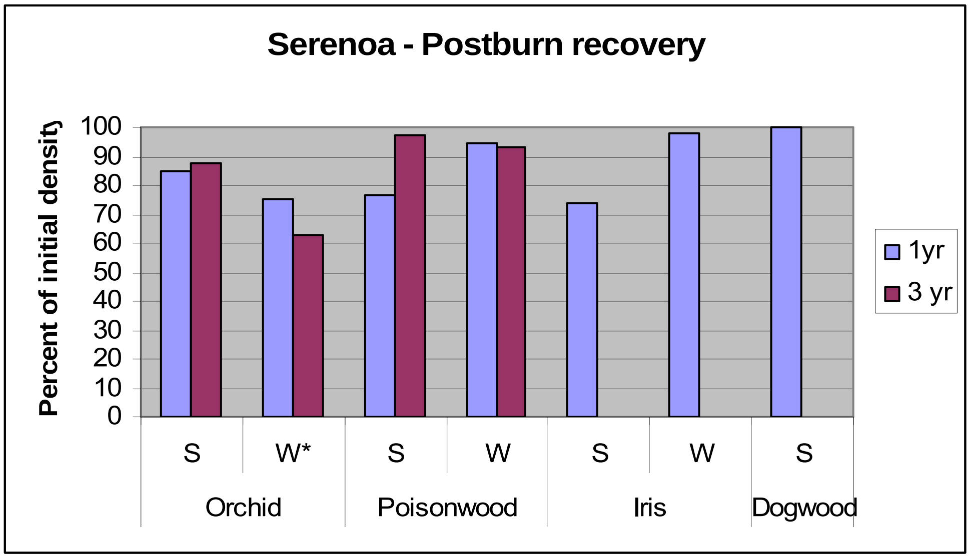


Figure 4.4. Number of palms with total heights ≥ 1 m one and three years after burns, expressed as percentage of number of palms with total heights ≥ 1 m before the burns. The Iris and Dogwood plots were not sampled 3 years after burning.
*The Orchid winter plot had < 10 individuals.

5. SHRUBS

Methods

Woody plants ≥ 1 m tall and with DBH < 5 cm were sampled in 20 circular subplots with radius of 4 m and an area of 50 m^2. The location of these shrub plots is shown in Figure 1.2. The species and height of each plant was recorded. For species with tree-like form (e.g., *Metopium*), the DBH was recorded if it was ≥ 1.0 cm. For species with shrubby growth form (e.g., *Myrica*), the maximum crown width and its perpendicular were measured. Shrubs were re-sampled at one-year intervals after the burns on the same schedule as palms: Orchid and Poisonwood plots were sampled a total of four times, the Iris and Dogwood control and summer burn plots were sampled three times, the Iris winter burn plot was sampled twice and the Dogwood winter and all the Locustberry and Buttonwood plots were sampled just once.

Results and Discussion

Initial Conditions

A total of 34 woody species were found in the shrub plots. *Pithecellobium guadelupense*, *Pisonia rotundata*, *Metopium toxiferum*, and *Psidium longipes* were found in all the study plots (Table 5.1). *Metopium* is the most common hardwood tree in the pinelands and the other three species are shrubs or small trees in the Florida Keys. Other common species included *Myrsine floridana*, *Byrsonima lucida*, and *Myrica cerifera*. All the species have tropical distributions with the exception of *Myrica*.

The density of hardwood stems was 2.5 times greater in the shrubby plots than the open plots (shrubby 5170±570, open 2050±230 stems/ha; t-test p = 0.0004). *Pithecellobium* was the dominant species in terms of stem density, accounting for over one third of all the stems (1315 stems/ha).

The number of shrub species found in shrubby plots was greater than found in the open plots (shrubby 14.4±1.17 species, open 10.3±1.15 species; t-test, p=0.024). The Iris control plot had unusually high species diversity for an open plot due to the presence of single individuals of five species. Conversely, the Dogwood winter burn plot had a low diversity for a shrubby plot.

Fire effects

The vast majority of shrub stems were topkilled by the fires, but all species are capable of regrowth from basal or belowground parts. All of the species can therefore be described as sprouters rather than seeders. One judge of the fire impact is the number of stems ≥ 1 m tall one year after the burn relative to the initial number of stems. Most of the stems recorded after the burns were resprouts from topkilled individuals, although there were instances where some stems, particularly larger stems or plants in areas of very low fuel loads, survived the fire. The percent recovery of initial stem number one year after burning is shown in Table 5.2 for the seven most common species. Species vary from having very few stems ≥ 1 m tall one year after burning, as in the case of *Myrsine*, *Pithecellobium*, and *Psidium*, to those species such as *Myrica* and *Pisonia* in which 30% or more of the initial number of stems are found a year later.

There is also variability among burns within species. Figure 5.1 compares the recovery of the three most common species after different seasons of burns. There are three pairs of winter/summer burns that can be compared in the Orchid, Poisonwood, and Iris sites. Our measures of fire severity (Table 2.3) indicated more severe fires in the summer than the winter in all three cases and summer fires are generally considered more stressful to hardwoods than winter fires (Robbins and Myers 1992). In spite of this, *Metopium* had a higher proportion of live stems after the summer burn in the Poisonwood site and *Pisonia* had more stems 1 year after summer burns at both Orchid and Poisonwood. The Dogwood summer burn (for which there was no paired winter burn) was the most stressful burn for four of the seven species (Table 5.2).

We are able to compare the recovery of stem numbers 3 years after burning in the paired plots at Orchid and Poisonwood (Table 5.3). *Byrsonima*, *Metopiu*m, and *Pisonia* all increased to reach about half the initial number of stems on average. The recovery of *Myrsine* and *Psidium* after fire is particularly slow. *Myrica*, on the other hand, because of prolific root sprouting actually had more stems 3 years after burning than before the fires.

The relative recovery of the three dominant species after winter and summer burns at the Orchid and Poisonwood sites is the same at 3 yr after burning as it was 1 yr postburn (Figure 5.2). The recovery of *Metopium* and *Pithecellobium* is less after the summer burn than the winter burn at the Orchid site, but at the Poisonwood site recovery was greater after the summer burns. *Pisonia* showed more vigorous recovery after summer burns at both sites. The number of stems of *Pithecellobium* increased dramatically between the first and third year after burning (from 1.3% of the initial number to 23.7%), although at three years postburn it was still considerably less recovered than four of the six other common species (Table 5.3). The increase between years was much more modest for *Metopium* (from 14.9 to 49.3) and *Pisonia* (from 28.4 to 54.9%).

Table 5.1. Shrub species diversity and number of stems > 1 m tall (per 1000 m^2) in the study plots before treatments.

	Orchid			Poisonwood			Iris			Dogwood			Locustberry			Buttonwood		
	OC	OS	OW	PC	PS	PW	IC	IS	IW	DC	DS	DW	LC	LS	LW	BC	BS	BW
No. of species	9	13	9	12	17	17	18	9	10	12	17	9	6	11	8	20	14	12
No. of stems	135	293	296	649	813	651	245	200	166	303	409	316	133	255	121	580	525	411
Species																		
Acacia pinetorum	2	1		3	1	1	2		1									
Annona glabra						1	6				3							
Byrsonima lucida	8	8	22	10	25	25	13	23	15	16	20	14		2	3	21	12	11
Coccoloba diversifolia				1		1	1											
Coccoloba uvifera					2	1										17	5	2
Conocarpus erectus		8	1	3	11	94		5								24		
Croton linearis				13	19	7			12						1			
Crossopetalum rhacoma		1					1			1	1					2	2	
Eugenia axillaris	4	1		8	9		9		15	4	4	9	3	5				
Eugenia foetida					10	5	40				1	1		1				
Ficus citrifolia							1			1	1			2				
Guettarda scabra		11	1							4	17	8			1			
Manilkara bahamensis					3	2										8	2	2
Metopium toxiferum	10	26	14	25	63	44	25	29	35	18	21	58	14	7	7	71	65	61
Morinda royoc	6	1																
Myrica cerifera		55	26	5	19	40	9	15	6	25	17	7		4	32	64	108	30
Myrsine floridana	7	38	71	51	4	203	42	89	37	5	22		4	37		57	58	50
Piscidia piscipula							2									3	1	1
Pisonia rotundata	17	34	37	59	70	32	31	14	26	57	42	47	30	36	20	44	31	27
Pithecellobium guadalupense	23	55	91	468	561	180	42	13	18	105	206	117	51	88	28	142	103	76
Psidium longipes	58	54	33	3	5	12	5	9	1	66	45	55	31	72	29	73	135	145
Randia aculeata					2	2	14	3		1	1					5		2
Reynosia septentrionalis							1				6			1		1	1	
Sophora tomentosa																26	1	4
Ximenia americana						1	1											

Other species: *Bumelia celastrina,* BC (4); *Chrysobalanus icaco* , DS (1); *Chrysophylum oliviforme* , DS (1); *Guapira discolor* , BC (10); *Jacquinia keyensis* , BC (1); *Rhizophora mangle* , BC (2); *Schinus terebinthifolius* (non-native), BS (1); *Simaruba glauca* , PS (3); Unid. Solanaceae, BC (5).

Table 5.2. Number of shrub stems >1 m tall one year after burning as percentage of initial number.

Site	Burn	Species						
		Byrsonima	Metopium	Myrica	Myrsine	Pisonia	Pithecell.	Psidium
Orchid	Summer	*	7.7	10.9	0.0	26.5	1.8	1.9
	Winter	9.1	14.3	34.6	0.0	16.2	2.2	0.0
Poisonwood	Summer	4.0	23.8	52.6	*	42.9	0.5	*
	Winter	12.0	13.6	45.0	3.9	28.1	0.6	8.3
Iris	Summer	13.0	0.0	13.3	0.0	21.4	0.0	*
	Winter	13.3	20.0	*	2.7	69.2	11.1	*
Dogwood	Summer	0.0	0.0	23.5	0.0	21.4	0.0	2.2
Mean		8.6	11.3	30.0	1.1	32.3	2.3	3.1

* <10 stems preburn

Table 5.3. Number of shrub stems >1 m tall three years after burning as percentage of initial number.

Site	Burn	Species						
		Byrsonima	Metopium	Myrica	Myrsine	Pisonia	Pithecell.	Psidium
Orchid	Summer	*	23.1	50.9	0.0	64.7	7.3	7.4
	Winter	40.9	50.0	100.0	4.2	40.5	23.1	3.0
Poisonwood	Summer	64.0	65.1	147.4	*	61.4	36.4	*
	Winter	44.0	59.1	145.0	34.0	53.1	27.8	8.3
Mean		49.6	49.3	110.8	12.7	54.9	23.7	6.2

* <10 stems preburn

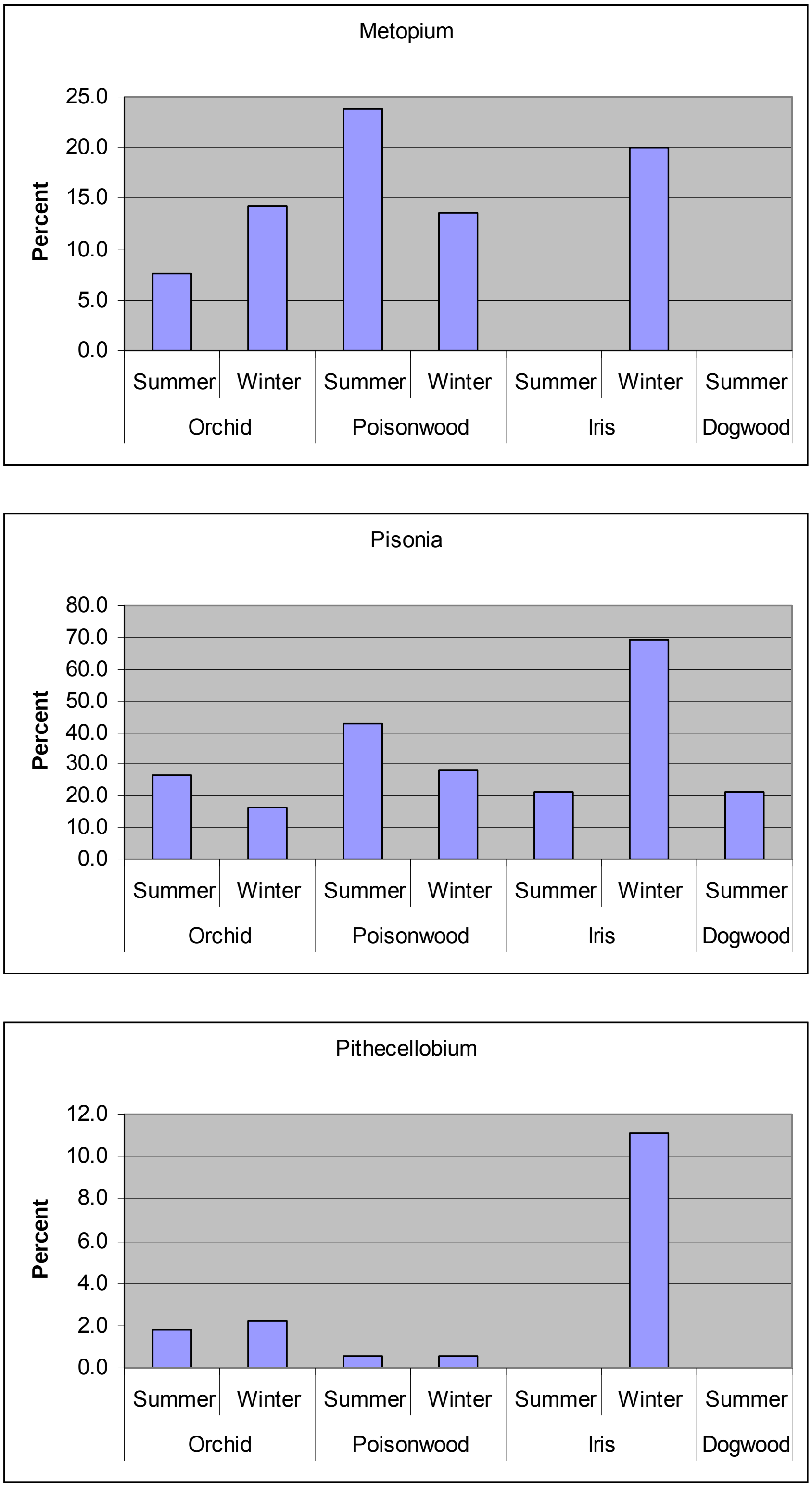


Figure 5.1. Hardwood stems >1m tall one year postburn as percent of initial number.

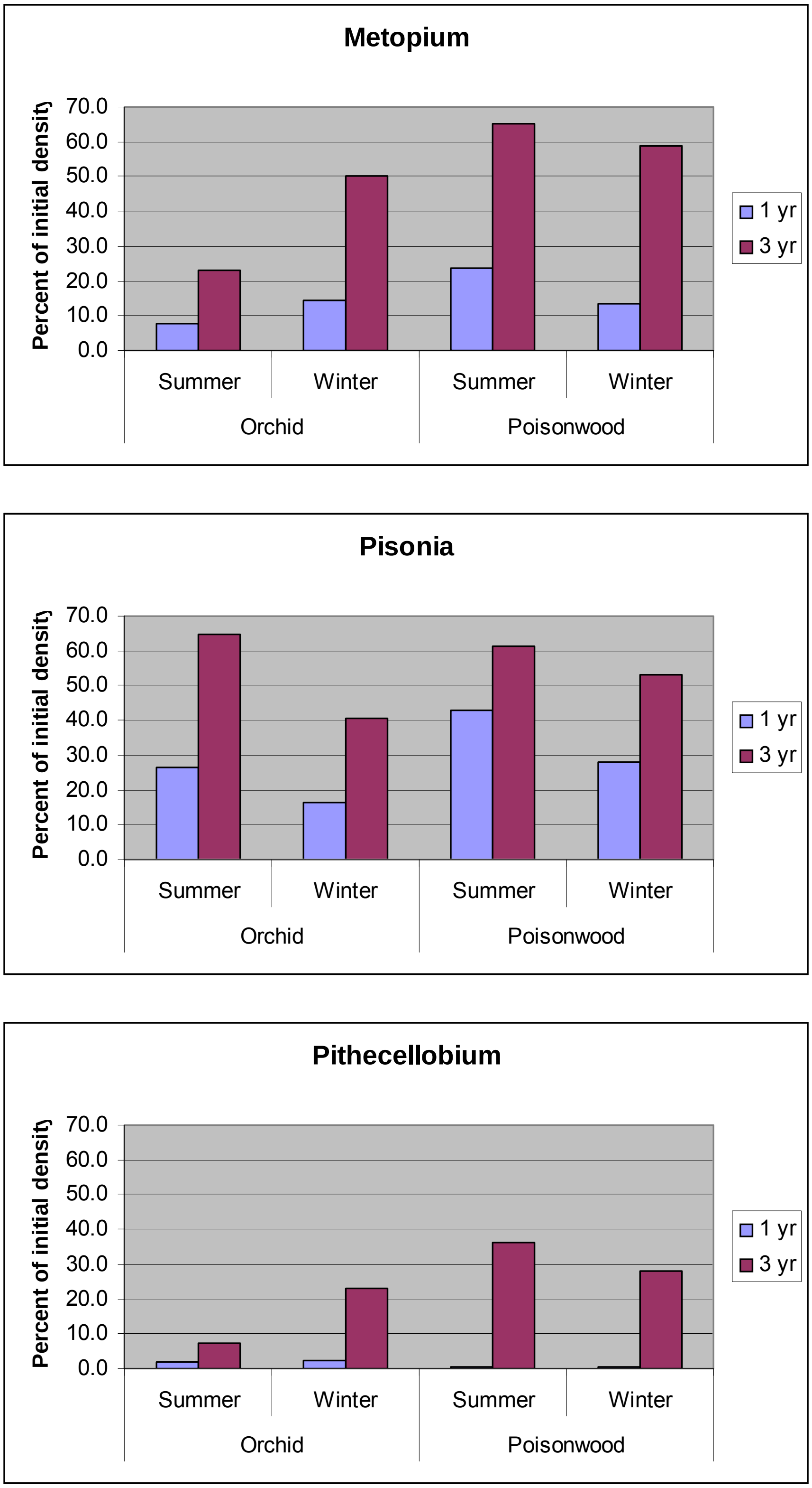


Figure 5.2. Comparison of one-year and three-year postburn recovery of hardwood stems.

6. HERB LAYER

Methods

We sampled the herb layer vegetation in 80 1.0-m^2 plots in each of the 18 treatment plots. Four circular plots were located within each of the 20 shrub plots, centered 2 m from the shrub plot center in the four cardinal compass headings. A large galvanized nail marked the herb plot center and we sampled in a 0.57 m radius around the center point for a sample area of 1.0 m^2. The herb layer consisted of all herbaceous plants as well as individuals of palms and woody species whose height was < 1.0 m.

Within each herb plot we listed each species present and estimated its cover according to the following cover classes: (1 = <1%, 2 = 1-5%, 3 = 5-10%, 4 = 10-25%, 5 = 25-50%, 6 = > 50%). Cover classes were converted to percent cover for analysis by using the midpoint of each range. The treatment plots were sampled shortly before the burns and at one-year intervals thereafter. The control plots were generally sampled at the same time as the summer burn plots. The Orchid and Poisonwood winter plots were sampled twice before burning and three years after burning. All sampling events are shown in Table 6.1. The December preburn samples in Orchid and Poisonwood winter plots were used in analyses presented here because subsequent sampling was also done in the winter.

We applied non-metric multidimensional scaling (NMS: McCune and Grace 2002), using the PC-ORD statistical package (McCune & Medford 1999), to characterize and visualize the variation in plant species composition among sites. NMS is an ordination method that arranges sites in a few dimensions, such that the distances among sites in ordination space match as closely as possible the rank order of their paired dissimilarities in species composition. An optimal solution is achieved via an iterative algorithm that reduces “stress” between the ordination and the matrix of site-by-site dissimilarities. We derived an estimate of dissimilarity among sites by applying the Bray-Curtis distance metric to untransformed species cover values. To reduce undue influence from very rare species, taxa that appeared in 5% of the plots or fewer were eliminated from the analysis. Ordinations were applied to two data sets. (1) To examine background variation in species composition across pine forests on Big Pine Key, we ordinated mean species cover for the 18 plots in their pre-burn condition. (2) To examine the temporal sequence in species composition associated with fire, we ordinated data from the 24 plot-by-year combinations from the Orchid and Poisonwood blocks during the years 1998-2001. In this case, rather than using stand means, we analyzed a data set in which mean species cover from the 20 subplots per stand (representing 4 quadrats each) were entered separately.

Use of subplot data, as described above, also provided the replication necessary to decompose the effects on species composition associated with Block (Poisonwood or Orchid), Burn Season (Summer, Winter, or Control) and Year (1998-2001) within a split plot design, using an analytical method developed by Anderson (2001, McArdle and Anderson 2001). The method is equivalent to a non-parametric ANOVA, and need not be based on a variance-covariance site matrix. Incorporation of a dissimilarity matrix based on Bray-Curtis distances, as we do here, eliminates problems associated with the undue influence of zero values and minor species in applications of MANOVA, for instance, to typical species-by-site data sets (Legendre and Anderson 1999).

Results and Discussion

Initial Conditions

A total of 163 taxa were identified in the herb plots during the course of the study: 5 ferns and fern allies, 77 species of forbs (including 14 vine species), 33 species of graminoids (grasses and sedges), 4 palms, and 44 woody species. Most of the woody species were smaller individuals of species found in the shrub and tree layers, although a few were small-statured woody species that rarely or never reach a meter in height (e.g., *Catesbaea parviflora*, *Crossopetalum ilicifolium*, and *Licania michauxii*). The best represented families in the herb layer were Poaceae (22 spp.), Asteraceae (17 spp.), Fabaceae (15 spp.), Cyperaceae (11 spp.), and Euphorbiaceae (11 spp.). The herb layer species diversity was higher in the plots in open sites than in the shrubby plots, with an average of about 14 more species present in the open plots (75.0 ± 3.00 species. in open vs. 61.4 ± 2.70 species. in shrubby plots, p=0.004).

The mean herb cover of the 18 plots before burning was 21.5%. Woody species contributed 9.3%, grasses and sedges 5.0%, forbs and ferns 4.5%, and palm seedlings 2.7%. As one would expect due to the competition from hardwood growth, the herb layer cover was greater in the open plots than the shrubby plots (24.7 ± 1.27% in open vs. 18.3 ± 1.59% in shrubby plots, p=0.006; Figure 6.1). Of the shrubby sites, Poisonwood had the highest cover of herbaceous species (Figure 6.1). Grasses were particularly prominent at the Iris site.

The NMS ordination of the pre-burn herb layer from all 18 plots is illustrated in Figure 6.2. The two-axis ordination represents the variability in the data set quite well, as indicated by a low “stress” value of 10.8. The clustering of plots within individual blocks is evident, indicating a considerable degree of spatial patterning within the Big Pine Key forest, and suggesting that our blocking layout was effective in capturing that variation. Plots in shrubby blocks (Poisonwood, Dogwood, and Buttonwood) generally occupy the upper half of the ordination, while open blocks (Iris, Orchid, and Locustberry) occupy the lower half. The ordination therefore reinforces the idea that development of the shrub layer affects the composition of the herb layer in these forests.

Fire effects

The impact of fire on the herb layer is relatively minor and short-lived, relative to impacts on trees and shrubs. The percent cover before and after fire in several plots is shown in Figure 6.3. There was a substantial decrease in herb layer cover the first year after the Orchid winter burn, but not in any of the other burns. The grasses showed a decrease in cover after burning in all four of the Orchid and Poisonwood plots, but by the third year the cover had returned to pre-burn levels. Normally, one would not expect it to take three years for grass cover to respond and it is not clear why this happened here. At the Iris site, the site with the greatest pre-burn graminoid cover, the cover returned to initial values within the year after burning.

Fire-associated changes in vegetation at the community level were addressed exclusively with data from the Orchid and Poisonwood blocks, where fires were successfully completed in the first year of the project, thereby enabling us to track vegetation composition from its pre-burn condition through the species assemblages that developed over the next three years. Analysis of these data is summarized in Table 6.2. The 3-way interaction is not a significant source of variation, but two of the 2-way interactions were significant, or nearly so: Block x Season and Season x Year. In indicating that the year-to-year compositional trajectories of the Control, Summer, and Winter burn treatments differed, the S x Y interaction is of particular interest.

The analysis of varance results can be examined visually in the NMS ordination diagram (Figure 6.4). As described earlier, the position of each plot-by-year combination in Figure 6.4 represents the centroid of 20 subplot-by-year positions. Time vectors are included in the figure to illustrate the developmental sequence within individual plots. As in the ordination of all pre-burn stands (Figure 6.2), “stress” is low (11.2), and plots from each block form distinct groups in ordination space. The near-significant S x Y interaction is manifested by more subtle differences in the temporal sequences among treatments. In the Control plots, ordination distances between successive years are usually short, indicating little change in composition, with the exception of 1998-99 in PC. We believe the 1998 position of this plot to have been strongly impacted by its January sampling date, when many of the herbaceous perennials had not fully emerged. Other than this segment, the Control plots trace closed circles in time. In comparison, the burned plots follow a series of trajectories, some (e.g., PS and PW) moving increasingly away from their point of origin in 1998, others (OS and OW) generally returning toward their 1998 position by Year 3 after fire, with relatively long distances between successive years. Nevertheless, it is clear that temporal changes due to fire were small in comparison to the among-plot differences in initial composition, and that a longer monitoring period is necessary to gain perspective on plant community changes following fire, which are likely to unfold over decades, not years.

Turnover in species composition in the herb layer resulting from the burns was minor. A few new species were recorded in plots after the initial sampling, but these were mostly single seedlings of woody species. One interesting case, however, is that of *Basiphyllaea corallicola*, a rare terrestrial orchid (Brown 2002). It is a small plant, at the most a few decimeters tall, with narrow, grass-like leaves. It occurred sporadically in a few of the plots, but it is unlikely that it was a response to burning. It was found in the IW plot one year after burning, the PW plot two years after burning, the OS plot 3 years after burning, and in the unburned IC plot in 2001 but not in previous years.

Jacquemontia pentanthos, skyblue clustervine, is a twining herbaceous member of the Convolvulaceae that exhibits fire-stimulated germination. We observed numerous seedlings in the months immediately following some of the burns. In the initial herb sampling, *Jacquemontia* was found only at single points in the Dogwood and Iris control plots. After burning it appeared at the Poisonwood site in dramatic fashion (Table 6.3). One year after the summer burn, *Jacquemontia* appeared in almost half the sampling locations and had a respectable cover of 1.6%. The response was not as strong after the winter burn, but pronounced nonetheless. The best explanation for this phenomenon is germination from a soil seed bank. The congeneric *J. curtisii* has been shown to have fire-stimulated germination in mainland South Florida slash pinelands (Spier and Snyder 1998). In that study, winter burns, which had higher temperatures, resulted in more germination than summer burns. It is possible that the difference in germination between seasons with *J. pentanthos* in Big Pine Key pinelands was due to differences in fire intensity, but we do not have sufficient data to draw that conclusion.

One herb species was studied in detail by a graduate student during the course of this study. Hong Liu looked at fire influences on *Chamaecrista lineata* var. *keyensis*, the endemic Big Pine partridge pea (Liu 2003). She found that *Chamaecrista* had higher mortality after summer than winter burns (Orchid, Poisonwood, and Iris sites) and suggested that this was due to either hotter fires in the summer or to inherent seasonal variation in plant vulnerability. It is considered unlikely that differences in fire temperatures between seasons is the primary explanation because no relationship between fire temperature and individual plant mortality was found (Liu et al. 2005).

Table 6.1. Herb layer sampling dates.

PLOT	**1998**					**1999**			**2000**			**2001**				
	START	**END**	**START**	**END**	**BURNED**	**START**	**END**	**BURNED**	**START**	**END**	**BURNED**	**START**	**END**	**START**	**END**	**BURNED**
OC	5-Mar	6-Mar				25-Aug	28-Aug		5-Sep	5-Sep		18-Oct	18-Oct			
OS	7-May	15-May			**16-Aug**	26-Aug	26-Aug		31-Aug	31-Aug		6-Sep	6-Sep			
OW	29-May	30-May	3-Dec	4-Dec	**15-Dec**	4-Nov	6-Nov		16-Nov	16-Nov		15-Nov	15-Nov			
PC	29-Jan	20-Feb				28-Aug	1-Sep		8-Sep	8-Sep		12-Oct	12-Oct			
PS	8-May	9-May			**17-Aug**	27-Aug	27-Aug		6-Sep	6-Sep		7-Sep	7-Sep			
PW	15-May	15-Jun	2-Dec	3-Dec	**15-Dec**	5-Nov	9-Nov		17-Nov	17-Nov		16-Nov	16-Nov			
IC						11-Jun	11-Jun		6-Jul	7-Jul		27-Sep	27-Sep			
IS						2-Jun	4-Jun	**14-Jul**	30-Jun	6-Jul		5-Jul	6-Jul			
IW						25-Jun	25-Jun		30-Nov	30-Nov	**12-Dec**	15-Nov	15-Nov			
DC						11-Jun	8-Jul		30-Jun	30-Jun		11-Jun	12-Jun	28-Sep	28-Sep	
DS						4-Jun	6-Jun	**22-Jun**	29-Jun	30-Jun		3-Jul	1-Oct			
DW						25-Jun	7-Jul		1-Dec	1-Dec		7-Jun	7-Jun			
LC									16-May	16-Jun		24-May	25-May			
LS									26-May	26-May		18-May	18-May			**19-Jul**
LW									16-May	16-Jun		24-May	24-May			**19-Jul**
BC									8-Jun	8-Jun		15-Jun	15-Jun			
BS									26-May	26-May		14-Jun	14-Jun			**18-Jul**
BW									7-Jun	8-Jun		13-Jun	14-Jun			**18-Jul**

Table 6.2. Non-parametric ANOVA on Bray-Curtis distances for species assemblages in two Blocks (Poisonwood and Orchid) undergoing three different Seasonal treatments (Control, Summer and Winter burning) during four years (1998, 1999, 2000 and 2001).

Source	df	SS	MS	F	P (MC)	Error term
Block (B)	1	145,337.76	145,337.76	56.69	0.01	Residual
Season (S)	2	71,122.71	35,561.36	1.33	0.20	B x S
B x S	2	53,457.21	26,728.60	10.42	0.001	Residual
Year (Y)	3	17,787.60	5,929.20	2.71	0.001	B x Y
B x Y	3	6,549.54	2,183.18	1.31	0.14	B x S x Y
S x Y	6	13,791.56	2,298.59	1.37	0.06	B x S x Y
B x S x Y	6	10,027.64	1,671.27	0.65	0.99	Residual
Residual	456	1,169,048.52	2,563.70			
Total	479	1,487,119.56				

Table 6.3. Frequency and mean cover of *Jacquemontia pentanthos* at the Poisonwood site before and after experimental burns. *Jacquemontia* did not appear in the control plot. Frequency based on number of shrub sampling points at which it occurred.

	Summer burn plot				Winter burn plot				
	Pre	1 yr post	2 yr post	3 yr post	Pre (May)	Pre (Dec.)	1 yr post	2 yr post	3 yr post
Frequency (%)	0	45	50	45	0	0	20	20	15
Cover (%)	0	1.62	1.43	0.86	0	0	0.06	0.15	0.08

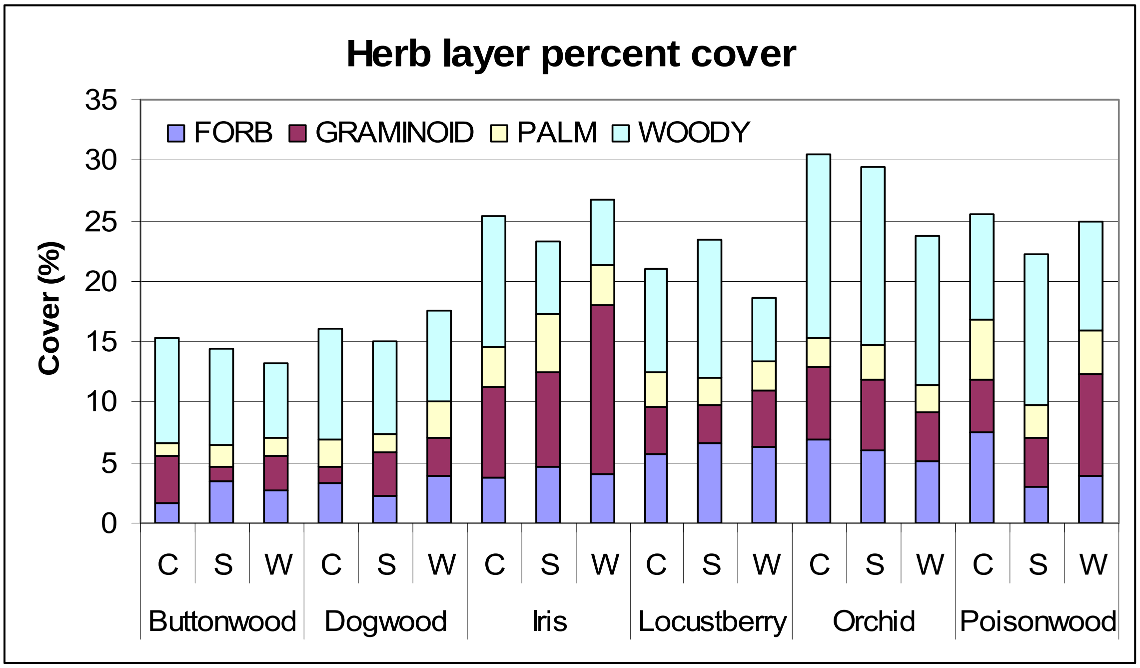


Figure 6.1. Mean herb layer cover in the 18 experimental plots before burns. The forb category includes ferns, graminoids are grasses and sedges. C= control, S= summer burn, and W= winter burn plots at the six study sites.

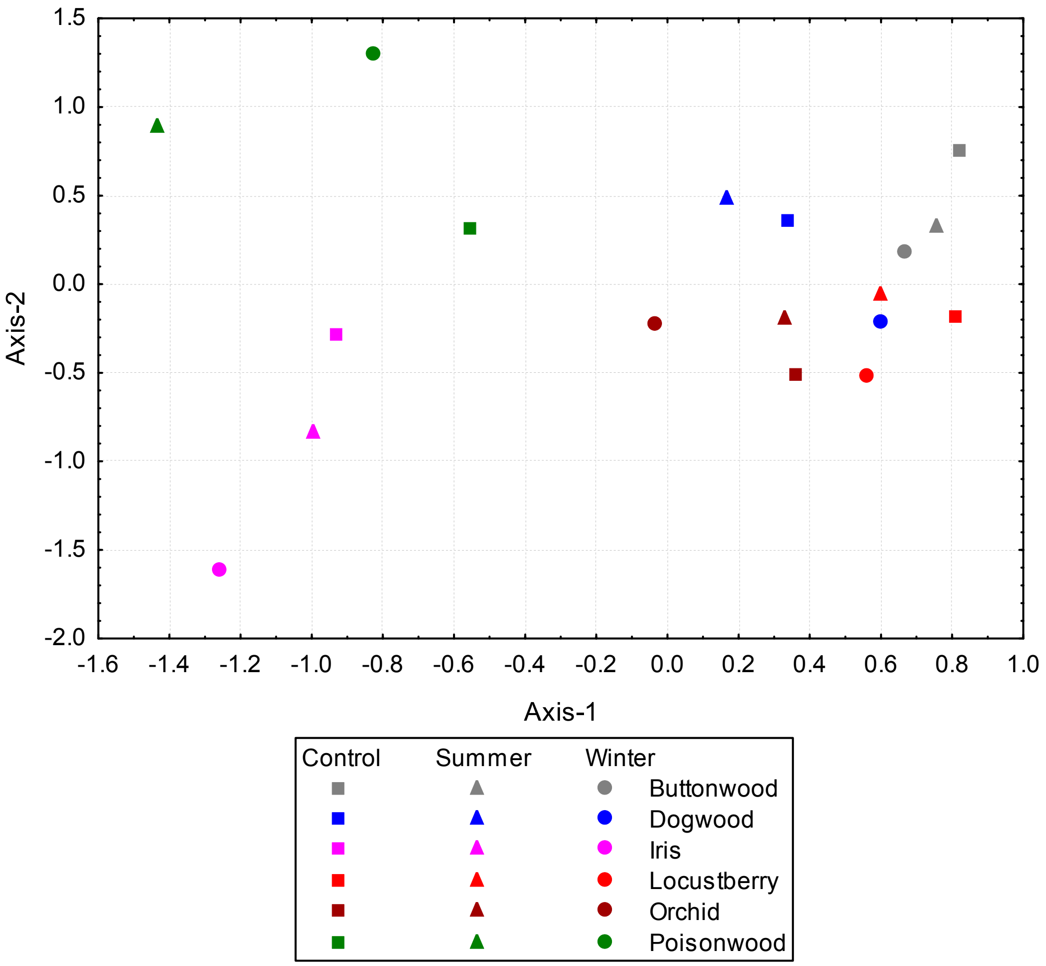


Figure 6.2. NMS ordination of pre-burn plots.

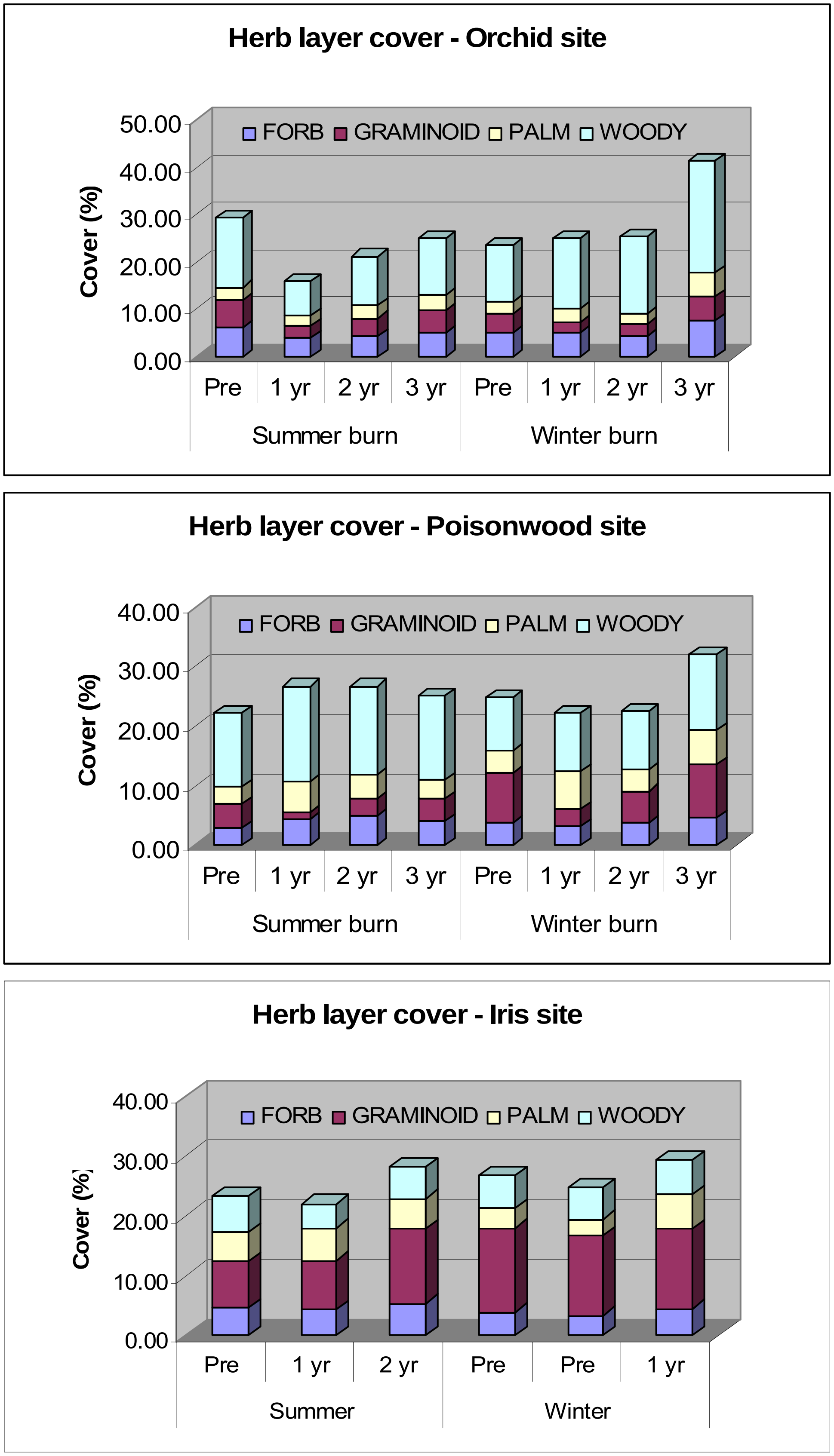


Figure 6.3. Mean herb layer cover over time. The Iris winter plot was sampled two years before and one year after the burn.

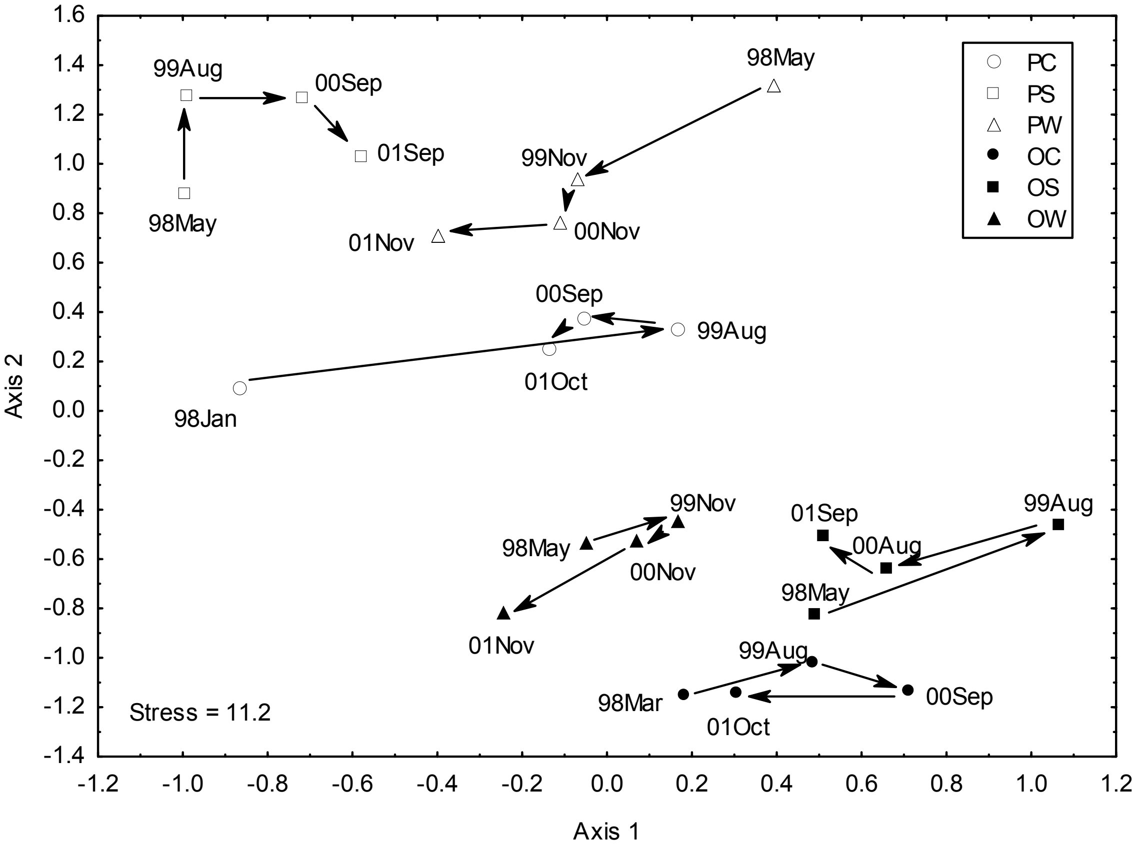


Figure 6.4. NMS ordination of Poisonwood (P) and Orchid (O) plots. The position of each plot-by-year combination represents the centroid of 20 subplot-by-year positions. The stress value is a measure of how well or poorly the distances in the ordination diagram correspond to the dissimilarity values calculated from the original data. It is generally accepted that a value below 15 represents a good fit.

7. HERBIVORY

Fire and mammalian herbivory have diverse consequences on vegetation via direct and indirect modification of bottom-up resources (Danell et al.1985). Consequences of both forces may affect plant architecture, phenology, and leaf quality, as well as the dynamics of the community. Fire and herbivory have been regarded as alternate, and sometimes similar, selection pressures (Paige 1992). The interaction of fire and herbivory has been studied in some systems, but few studies have looked at the whole plant community.

Season of burning can have marked influence on the response of pine forest plants (Platt et al. 1988, Snyder 1986, Spier and Snyder 1998, Waldrop et al. 1992), and there is some evidence that season of burn can affect the level of herbivory on certain plants. White-tailed deer ate more buds of *Pityopsis graminifolia* following January fires than growing season fires, although shoot production of this species was greatest following May burns, the time when the longleaf pine savannahs are most likely to burn (Brewer and Platt 1994). Seasonal movements of Key deer influenced by the availability of foods and fresh water may result in differences in damage to plants with different reproductive phenologies experiencing different seasonal burn regimes.

Fire and Key Deer Habitat

The importance of fire for Key deer habitat maintenance is well known (Dickson 1955, Alexander and Dickson 1972, Carlson 1989, Carlson et al. 1993). Robertson (1955) studied the vegetation of the area as part of an analysis of South Florida breeding birds. Recent studies (Ross et al.1992a,b; Rocca 1997) document the physical factors associated with the distribution of Lower Keys pinelands. Fire is essential in maintaining pine rockland as a pine-dominated forest; without periodic fires, succession proceeds to hardwood hammock with a relict overstory of pine (Robertson 1955, Alexander 1967, Snyder et al. 1990).

The Key deer, *Odocoileus virginianus clavium*, is the smallest subspecies of the North American white-tailed deer. The Key deer is both federally and state listed as endangered due to the continual reduction of its habitat and changes in the quality of its habitat due to human influences. Key deer utilize a variety of habitats within their range; pine rocklands are important to Key deer because they are sites with freshwater sources essential to their existence (Folk et al. 1991). Although mangroves constitute a large percentage of the diet of Key deer, many of their other preferred food plants (such as *Pithecellobium keyense, Acacia pinetorum, Morinda royoc*, and *Stylosanthes hamata*) are found in pine rockland habitat (Klimstra and Dooley 1990). A wide variety of pine rockland plants are utilized by Key deer to at least a moderate extent (Dooley 1975).

Pine rockland plant production is stimulated by fire, and plants resprouting after fire may be exceptionally palatable to browsing herbivores, with soft new shoots and leaves. Some woody species show significant increases in nutritive value following fire, though these changes only last a matter of months (Carlson 1989). The diet of the Key deer varies seasonally, and availability of special foods influences Key deer movements (Carlson et al. 1993; Klimstra and Dooley 1990). Controlled burning of pine rockland habitat can improve habitat conditions for Key deer and may have positive effects on population size (Klimstra 1986). Carlson (1989) monitored regrowth of seven woody species and browsing by Key deer on those species in burned and unburned areas of the Refuge, and found that those woody species important in the diet of the deer recover within two years of the fire. He reported that herbaceous species responded quickly, with little change in species richness or frequency of occurrence of important diet and endemic species. This comparison was between burned (wet season) and unburned pine rockland sites, and did not involve deer exclusion. An earlier, more limited study (Dickson 1955) came to a similar conclusion for several species studied. Our experimental study provides some real data on this important subject, especially at a time when Key deer populations are at an all-time high.

Hypotheses

Hypothesis 1 – Key deer shape the pine rockland plant community via selective herbivory.

Hypothesis 2 – Perennial plants preferred by Key deer will achieve greater coverage in exclosures.

Hypothesis 3 – The season of burn will affect how a plant species compensates for tissue lost to deer herbivory.

Hypothesis 4 – The longer deer are excluded from sites, the greater the coverage of preferred perennials. Woody species may eventually outcompete shorter herbaceous species.

Hypothesis 5 – The interaction of fire (season of burn) and deer herbivory will have a significant effect on plant community composition in Lower Keys pine rockland habitat.

Methods

After the first-year treatment fires, while monitoring herb and shrub plots, we noticed many signs of deer activity and herbivory. We took advantage of the second-year experimental burns to study the effects of Key deer herbivory on regeneration of the understory pine rockland plant community after fire, contrasting burns in different seasons with unburned control areas.

In two blocks (Dogwood and Iris) we had three burn units: 1999 summer burn, 1999 winter burn, and control (unburned). These two blocks differed in their time-since-burn and openness (Dogwood was considerably more overgrown and had not been burned in 11 years; Iris block was more open, and had been burned either 8 (IW) or 14 (IC and IS) years earlier.

The first small exclosures were erected July and August (1999) following the summer burns in June and July 1999, in summer burn and control units. In only one of the blocks (Iris) did the winter burn take place as planned. The Dogwood winter burn was not completed within the framework of this project, so data for winter burn effects represent only one block.

Within each burn unit (1 ha), twenty shrub plots (4 m in diameter) had previously been located stratified-randomly and four herb plots (1 m^2) were nested within each of the twenty shrub plots for the main fire and vegetation study. The remaining twelve plots in the stratified array (Figure 1.2a) were used for this Key deer herbivory study, with two of the four herb plots fenced and two left open within each shrub plot. The central points of each plot were marked with rebar, using a wire to mark the radius of the circular plot when sampling.

Small exclosures (circles of 4 ft high field fence, each with a diameter of ca 1.5 m, surrounding 1 m^2 circular plots with diameters of 1.2 m) were erected. Two exclosures were placed at opposite cardinal directions 4 m from the central point of the shrub plot, and 2 non-fenced control plots at the other cardinal directions. Exclosures were sturdy (staked with 4 rebar) yet easy to open for sampling (the edges overlapped, with plastic covered heavy wire loops as latches). Most importantly, they were effective at keeping Key deer out of the herb plots contained within. Construction of exclosures in pine rockland was challenging because most rebar posts must be placed in solid rock using a heavy-duty hammer drill with attendant power source (a gas-powered generator in a wheelbarrow).

We sampled understory vegetation using methods similar to those used in our fire study (reported elsewhere), to facilitate comparison inside and outside the exclosures. Within each burn unit and the unburned control unit, 24 herb plots inside exclosures and 24 unfenced plots were sampled. We sampled all plants encountered in our plots (171 taxa total) when measuring abundance and cover of shrubs and herbs (Table 7.1). For monitoring plant height and herbivory we tagged individuals of 33 focal taxa (Table 7.1), of which only 20 were represented by more than 10 individuals over all the plots. These focal taxa were chosen to represent both plants that Key Deer had been observed to eat and species that appeared unpalatable.

In each plot, we collected the standard herb data as in the fire project, listing all species (< 1 m tall) present in the plot, counting the number of individuals or ramets, and estimating cover using a scale (1 = <1%, 2 = 1-5%, 3 = 5-10%, 4 = 10-25%, 5 = 25-50%, 6 = > 50%). For focal species (Table 7.1) we tagged up to five individual plants with bird bands (yellow bands inside the exclosures, green bands in unfenced control plots) and measured plant height (the longest stem length), number of stems, number of flowers/ inflorescences (or fruits/infructescences), and recorded evidence of herbivory by deer and other herbivores. We sampled plots 6 months and one year after fire treatments and fencing.

The omission of the winter fire from the Dogwood block gave us an unbalanced design, but for comparisons of stem length, density, and cover, data were combined for both blocks as blocks were not significantly different. However, open pine rockland sites (Iris block) have greater species diversity than overgrown sites (Dogwood block), so comparisons of diversity were made only for Iris block, as combining data would have misrepresented potential differences between fire treatments.

Univariate Analysis of Variance (ANOVA) was used to evaluate effects of fire season (burn) and treatment (fenced vs. open) on plant height, density, and cover of individual species, and species diversity of the community. Two non-metric multivariate analyses, analysis of similarity (ANOSIM) and non-metric multi-dimensional scaling (MDS) were carried out on species abundance and cover matrices using PRIMER 5.2.9. (Clarke and Gorley 2001). Species abundance and cover were scaled up to the sub-plot (or shrub plot) level. Variable "site" was created to consolidate block and fire treatment. Since ANOSIM can only analyze two independent factors at a time, we first examined the effects of site and exclosure, then the effects of block and fire treatment. The procedure generates a plot in which the separation between groups can be visualized, and higher values of the statistic, R, indicate better separation between groups.

Results and Discussion

We recorded a great deal of damage to plants in the burned sites, both in non-excluded plots, and in the buffer zone around the herb plots just inside the exclosures (deer muzzles can reach in several centimeters through the fencing). Evidence of deer activity was greater in burned areas where plants were regrowing, though we also observed deer damage on some plants in unburned control areas.

Most forbs with an erect flowering stem are subject to clipping by Key deer, and we observed many individuals in burned and control areas with upright stems removed. Most stems (> 60% of individuals in burned areas) of the endemic *Chamaecrista keyensis* were nipped off several centimeters from the ground if the plants are in the open, whereas the same species in the exclosures are undamaged by deer. Leaf-tying caterpillars appeared to damage both groups equally, but these effects were not quantified. *Aletris lutea, Pinguicula pumila, Pityopsis graminifolia, Polygala boykinii*, *Ruellia caroliniana,* and *Vernonia blodgetii* all experienced clipping by deer, though these taxa, with the exception of *Ruellia,* were not abundant enough in our experimental plots to get tagged and monitored individually.

The herb layer is not the only stratum affected by deer herbivory. Deer seemed to especially feed on new leaves and flowers produced by growing shoots. Seedlings, as well as shoots on saplings, of woody *Pithecellobium guadalupense* were eaten by Key deer outside exclosures. Key deer also fed upon new leaves of silver palm (*Coccothrinax argentata*) and thatch palm (*Thrinax morrissii*). Leaves and meristems were damaged significantly more when plants were outside exclosures. Species morphologically armed for protection were also eaten by Key deer. We have seen adults carefully eat the tender leaves of prickly *Acacia pinetorum,* and there are many individuals of thistles (*Cirsium horridulum*) with basal rosettes missing. The endemic *Catesbya parviflora* (Rubiaceae), a very spiny shrub of short stature, was often seen with browsed stems.

A comparison of stem lengths of all species showed significant interaction of both fire and exclosure (p = .006, Table 7.2). Mean stem length was slightly lower within exclosures for plots in unburned control units, but for both winter and summer burn units, stem length was substantially greater within exclosures than in open plots (Figure 7.1).

Stem length was substantially greater for several twining leguminous species inside the exclosures (*Galactia parvifolia, Rhynchosia parviflora*, and *Stylosanthes hamata*) than for counterparts in the open (Figure 7.2). Some other vines appeared to be protected from deer browsing: *Smilax havanensis* (Liliaceae) showed no difference in control areas, yet vines were much longer in exclosures in both summer and winter burn units; but no differences were seen in *Mikania scandens* (Asteraceae), *Morinda royoc* (Rubiaceae), and *Jacquemontia pentanthos* (Convolvulaceae).

Density (number of individuals per m^2) showed no effect of exclosure, but a nearly significant ($p = .08$) effect of burn (Table 7.3). It is surprising that more individuals are present after winter burns than summer burns or controls (Figure 7.3), but since cover is less (see below) perhaps there is more room for establishment of new seedlings and ramets.

Univariate analysis of cover similarly shows no significant effect of exclosure, but a significant effect of burn (Table 7.4). Cover is lower in burn units than unburned controls after one year (though summer burn unit plots nearly recover fully compared with fenced unburned control plots), but interestingly the open plots in both summer and winter burn have equivalent cover. Exclosures seem to make a big difference after summer burn, with fenced plots showing substantially greater cover than open plots (Figure 7.4) but the differences are not statistically significant.

Species richness can be compared within Iris block only, and treatment effects appear to be significant ($p = 0.016$, Table 7.5): There is no significant interaction between burn and treatment, and for summer burn, winter burn, and unburned control plots, the mean number of species is greater in exclosure plots (Figure 7.5). The greatest mean number of species were found after summer burns in exclosures.

Using multivariate techniques, only block and fire treatment effects were significant, while exclosure effects were not. This pattern was reflected in values of R, and to some degree P values of the ANOSIM statistics (Table 7.6). Specifically, factors of site, block and fire treatment had much higher R values than exclosure. Higher R values indicate better separation between the groups. Results of MDS agreed with that of ANOSIM.

While many other studies have focused on deer use of plants and the effects of plants on deer movement, none have measured, with controls, the effects that Key deer have on the vegetation, and on specific plant taxa. Fire intensifies deer interest in plants and herbivory in its aftermath, and our design allowed us to detect short-term effects of deer grazing on plant establishment, growth, and the composition of the plant community. The interaction of fire and herbivory reveals some surprising things about plant community and population dynamics in pine rockland habitat.

The Key deer now enjoy a protected existence as an endangered species inhabiting the Key Deer Refuge, but their major perils are still related to the presence of human beings (vehicular traffic and loss of habitat). The deer may have been over-protected with respect to excluding fire from the area. The small-scale fires employed in our experiment, coupled with Key deer numbers at an all-time high, may exaggerate the effects on the small areas full of luxuriant regrowth after fire. Continuing studies of this type will be important as fire management involves more frequent fires of larger extent.

Table 7.1 – Plant species of pine rockland habitat encountered in this study. Species of special interest (individual stems tagged in plots) indicated in **bold type**. * indicates rare species. # indicates species reported or previously observed to be eaten by Key deer. Burn units are: DC = Dogwood Control, DS = Dogwood Summer Burn [Dogwood Block did not receive a winter burn]; IC = Iris Control, IS = Iris Summer, IW = Iris Winter.

Scientific name	**Family**	**Burn Units occurring**	**No. tagged**
Abildgaardia ovata	Cyperaceae	IC, IS, IW	10
#Acacia pinetorum	Fabaceae	IS, DS	0
Acalypha chamaedrifolia	Euphorbiaceae	IC, IS, IW	4
Agalinis keyensis	Scrophulariaceae	DC, DS, IC, IS, IW (all)	36
#Aletris bracteata	Liliaceae	IS	1
Andropogon cabanisii	Poaceae	DC, DS, IC, IW	5
Andropogon glomeratus	Poaceae	DC, IS, IW	3
Andropogon virginicus	Poaceae	IS, IW	0
Anemia adiantifolia	Schizaeaceae	DC, DS, IC, IS, IW (all)	11
Angadenia berteri	Apocynaceae	DC	0
Aristida purpurascens	Poaceae	DC, DS, IC, IS, IW (all)	3
Aster dumosus	Asteraceae	IC, IS, IW	4
Aster tenuifolius	Asteraceae	IW	0
Bletia purpurea	Orchidaceae	IS, IW	1
Borreria terminalis	Rubiaceae	DC, DS, IS, IW	2
Bumelia salicifolia	Sapotaceae	DS	1
Byrsonima lucida	Malpighiaceae	DC, DS, IC, IS, IW (all)	43
Cassia bahamensis	Fabaceae	IC	4
Cassytha filiformis	Lauraceae	DC, DS, IC, IS, IW (all)	1
Centrosema virginianum	Fabaceae	DC, DS, IC, IS, IW (all)	47
Chamaecrista aspera	Fabaceae	IS, IW	2
****#Chamaecrista lineata***	Fabaceae	DC, DS, IC, IS, IW (all)	84
****Chamaesyce adenoptera***	Euphorbiaceae	DS, IC, IS, IW	5
****Chamaesyce deltoidea***	Euphorbiaceae	DC, IC	9
Chiococca pinetorum	Rubiaceae	DC, DS, IC, IS, IW (all)	12
Cirsium horridulum	Asteraceae	DC, IC, IS, IW	2
Cladium jamaicense	Cyperaceae	IC, IS, IW	4

Scientific name	**Family**	**Burn Units occurring**	**No. tagged**
Cnidoscolus stimulosus	Euphorbiaceae	IW	0
Coccoloba diversifolia	Polygonaceae	DC	1
Coccoloba uvifera	Polygonaceae	DC, DS, IS	3
Scientific name	**Family**	**Burn Units occurring**	**No. tagged**
Coccothrinax argentata	Arecaceae	DC, DS, IC, IS, IW (all)	50
Crossopetalum ilicifolium	Celastraceae	IW	0
Crossopetalum rhacoma	Celastraceae	IC, IS	8
Crotolaria maritima	Fabaceae	DC, IC, IS, IW	4
#Croton linearis	Euphorbiaceae	DC, DS, IC, IS, IW (all)	73
Cynanchum blodgettii	Asclepiadaceae	IC, IW	0
Cynoctonum sessilifolium	Loganiaceae	DC, IC, IS, IW	7
Dichanthelium dichotomum	Poaceae	IS	0
****Dichromena floridensis***	Cyperaceae	DC, DS, IC, IS, IW (all)	111
Dyschoriste oblongifolia	Acanthaceae	IS	2
Echites umbellata	Apocynaceae	DC, DS, IC, IS, IW (all)	2
Eleocharis caribaea	Cyperaceae	IS	0
Eragrostis elliottii	Poaceae	IC, IW	0
Ernodea littoralis	Rubiaceae	DC, DS, IC	5
Eugenia axillaris	Myrtaceae	IC	1
Eugenia foetida	Myrtaceae	IC	1
Eustachys petraea	Poaceae	IC, IS, IW	5
Evolvulus sericeus	Convolvulaceae	DC, IC, IS, IW	15
Fimbristylis castanea	Cyperaceae	DC	0
Flaveria linearis	Asteraceae	IC, IS, IW	1
#Galactia parvifolia	Fabaceae	DC, DS, IC, IS, IW (all)	58
Guettarda scabra	Rubiaceae	DC, DS	10
Habenaria quinqueseta	Orchidaceae	DC	1
Hedyotis nigricans	Rubiaceae	DC, DS, IC, IS	3
#Heterotheca graminifolia	Asteraceae	DC, DS, IC, IS	4
Hypoxis wrightii	Hypoxidaceae	DC, DS, IC, IS, IW (all)	14
Indigofera miniata	Fabaceae	IS	0
Jacquemontia pentanthos	Convolvulaceae	DS, IC	4
Liatris laevigata	Asteraceae	DC, DS, IC, IS, IW (all)	11
Manilkara bahamensis	Sapotaceae	DC	1
****Melanthera parvifolia***	Asteraceae	IS, IW	13

Scientific name	**Family**	**Burn Units occurring**	**No. tagged**
Metopium toxiferum	Anacardiaceae	DC, DS, IC, IS, IW (all)	20
Mikania scandens	Asteraceae	IS, IW	4
#Morinda royoc	Rubiaceae	DC, DS, IC, IS, IW (all)	127
Muhlenbergia filipes	Poaceae	IS, IW	0
Myrica cerifera	Myricaceae	DC, DS, IC, IS	10
Scientific name	**Family**	**Burn Units occurring**	**No. tagged**
Myrsine floridana	Myrsinaceae	DC, DS, IC, IS, IW (all)	15
Panicum virgatum	Poaceae	DS, IC, IS, IW	0
Paspalum sp.	Poaceae	DC, DS, IC, IS, IW (all)	5
Phyllanthus pentaphyllus	Euphorbiaceae	DC, DS, IC, IS, IW (all)	18
Pinus elliottii	Pinaceae	DC, IC, IS, IW	12
Pinguicula pumila	Lentibulariaceae	IS, IW	1
Piriqueta caroliniana	Turneraceae	DC, DS, IC, IS, IW (all)	26
Piscidia piscipula	Fabaceae	IC	0
Pisonia rotundata	Nyctaginaceae	DC, DS, IC, IS, IW (all)	48
#Pithecellobium guadalupense	Fabaceae	DC, DS, IC, IS, IW (all)	60
Pluchea rosea	Asteraceae	IW	0
Polygala boykinii	Polygalaceae	DS, IS, IW	2
Polygala grandiflora	Polygalaceae	DC, DS, IC, IS, IW (all)	4
Psidium longipes	Myrtaceae	DC, DS, IC, IS	6
Pteridium aquilinum	Dennstaedtiaceae	DS	0
Pteris bahamensis	Adiantaceae	DC, DS, IC, IS	4
Pterocaulon pycnostachyum	Asteraceae	DC	1
Randia aculeata	Rubiaceae	DC, DS, IC, IS, IW (all)	22
Reynosia septentrionalis	Rhamnaceae	DS, IC	3
Rhynchosia parvifolia	Fabaceae	DC, DS, IC, IS, IW (all)	17
#Ruellia caroliniensis	Acanthaceae	DC, DS, IC, IS, IW (all)	55
Sabal palmetto	Arecaceae	IC, IS, IW	3
Sachsia polycephala	Asteraceae	IS, IW	7
Samolus ebracteatus	Primulaceae	IW	0

Scientific name	**Family**	**Burn Units occurring**	**No. tagged**
Schizachyrium gracile	Asteraceae	DC, DS, IC, IS, IW (all)	3
Schizachyrium rhizomatum	Asteraceae	IC, IS, IW	4
Schizachyrium semiberbe	Asteraceae	DC, DS, IC, IS, IW (all)	3
Schoenus nigricans	Cyperaceae	IW	0
Scleria verticillata	Cyperaceae	IS, IW	4
Serenoa repens	Arecaceae	IS, IW	33
Sisyrinchium miamiense	Iridaceae	DC, DS, IC, IS, IW (all)	0
#Smilax havanensis	Smilacaceae	DC, DS, IC, IS, IW (all)	102
Scientific name	**Family**	**Burn Units occurring**	**No. tagged**
Solidago stricta	Asteraceae	IC, IS, IW	3
Sophora tomentosa	Fabaceae	DS	1
Sorghastrum secundum	Asteraceae	DC, DS, IC, IS, IW (all)	88
*#_Stylosanthes hamata_	Fabaceae	IC, IW	3
Thrinax morrisii	Arecaceae	DC, DS, IC, IS, IW (all)	44
Tragia saxicola	Euphorbiaceae	DC, DS, IC	7
#Vernonia blodgettii	Asteraceae	DC, IC, IS, IW	2
Ximenia americana	Olacaceae	IW	1

Table 7.2. Univariate Analysis of Variance of mean stem length for all species one year post-burn. TRT = treatment, O = open (unfenced), X = exclosure (fenced), BURN = season of fire (control, summer, winter).

Source	df	F	p	power
Model	5	6.04	< ,0001	0.996
TRT (O,X)	1	9.31	0.002	0.862
BURN (C,S,W)	2	5.42	0.004	0.847
TRT x BURN	2	5.20	0.006*	0.831

Table 7.3. Univariate ANOVA for Density of Selected Species at 1 yr. Sp = species, other abbreviations as in Table 7.2.

Source	df	MS	F	p
Model	272	13.5	1.66	< 0.0001
Species	93	18.1	2.21	< 0.0001
TRT	1	4.1	0.51	0.477
Burn	2	20.6	2.53	<u>0.080</u>
Sp * TRT	57	5.5	0.68	0.968
Sp * Burn	71	9.8	1.2	0.128
TRT * Burn	2	.5	0.06	0.944
Sp*TRT*Burn	43	3.7	0.46	0.999

Table 7.4. Univariate ANOVA for Cover of all species at 1 yr

Source	df	MS	F	p
Model	5	1.3	1.70	0.131
TRT	1	0.4	0.52	0.47
Burn	2	2.1	2.90	0.05 *
TRT * Burn	2	0.54	0.73	0.50

Table 7.5 – Univariate ANOVA for species richness (Iris Block only) at 1 yr

Source	df	MS	F	p
Model	5	158.8	1.65	0.151
Burn	2	68.8	0.71	0.491
TRT	1	572.0	5.95	0.016
Burn * TRT	2	42.2	0.44	0.646

Table 7.6. Statistics from Two-way crossed ANOSIM.

factor	Based on plant cover		Based on plant abundance	
	R	P	R	P
site	0.481	**0.001**	0.353	**0.001**
exclosure	0.103	**0.001**	0.025	0.146
Block	0.653	**0.001**	0.442	**0.001**
Fire treatment	0.36	**0.001**	0.189	**0.001**

Figure 7.1. Mean stem length of selected species one year after treatment + S.E. all spp.

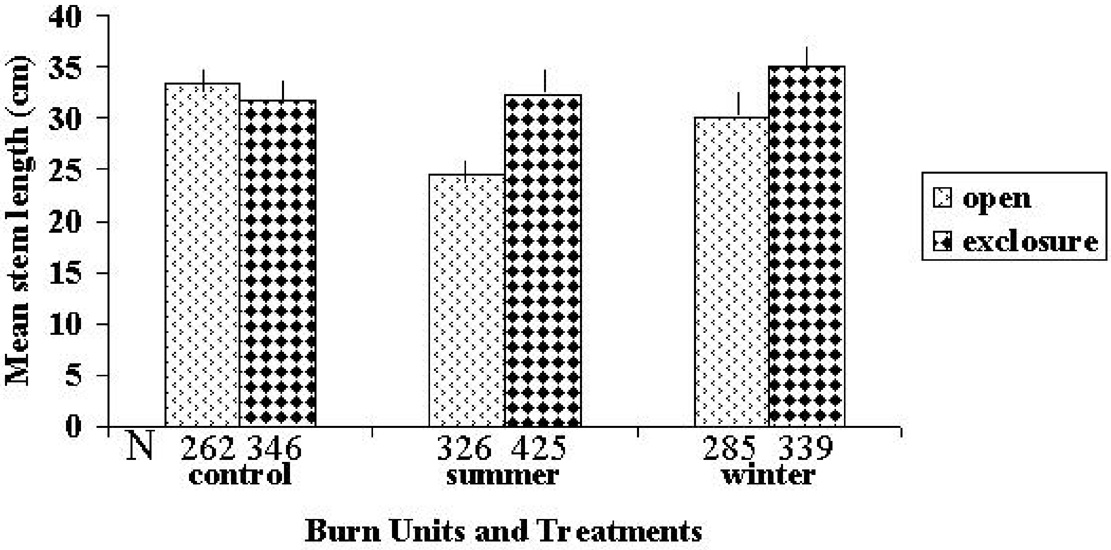
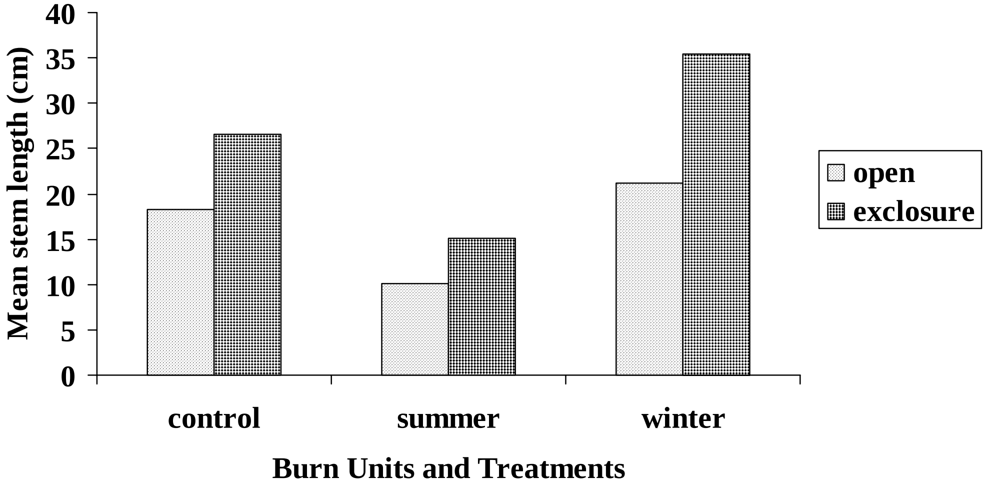
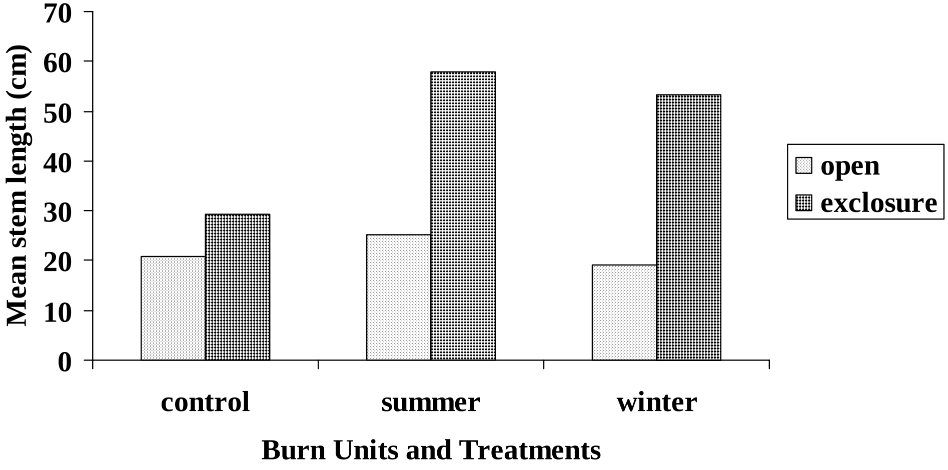


Figure 7.2. Mean stem length of leguminous vines one year after treatment.

a) *Centrosema virginiana*



b) *Galactia parvifolia*



c) *Rhynchosia parviflora*

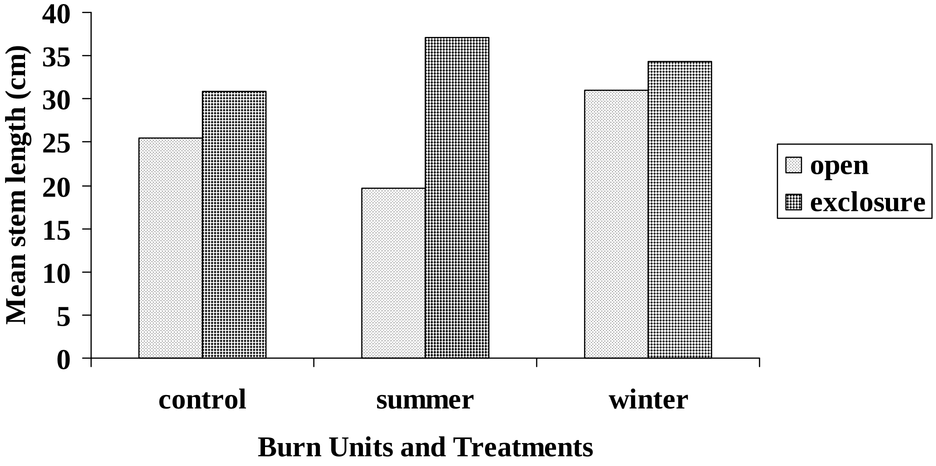


Figure 7.3. Mean Density of Selected Species one year after treatment + S.E.

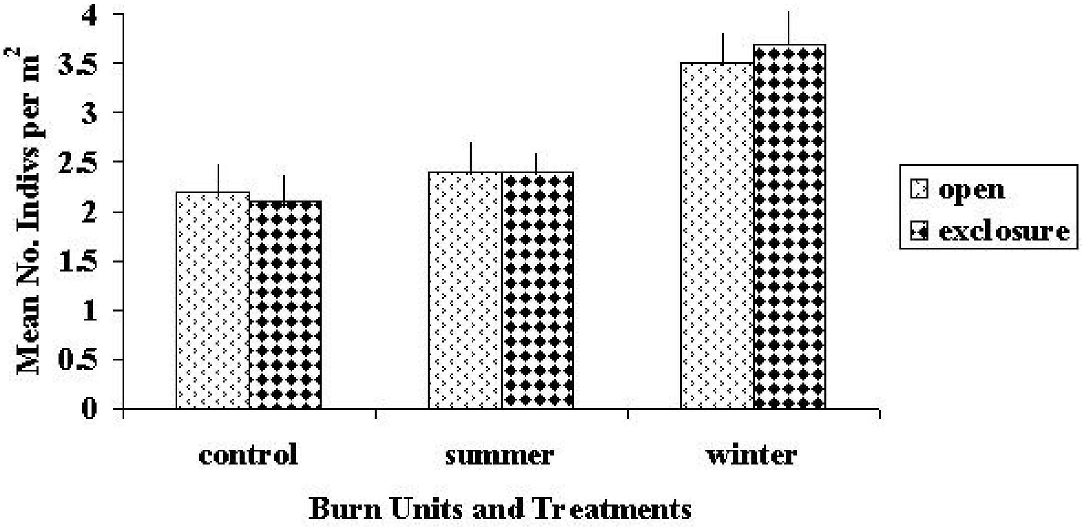
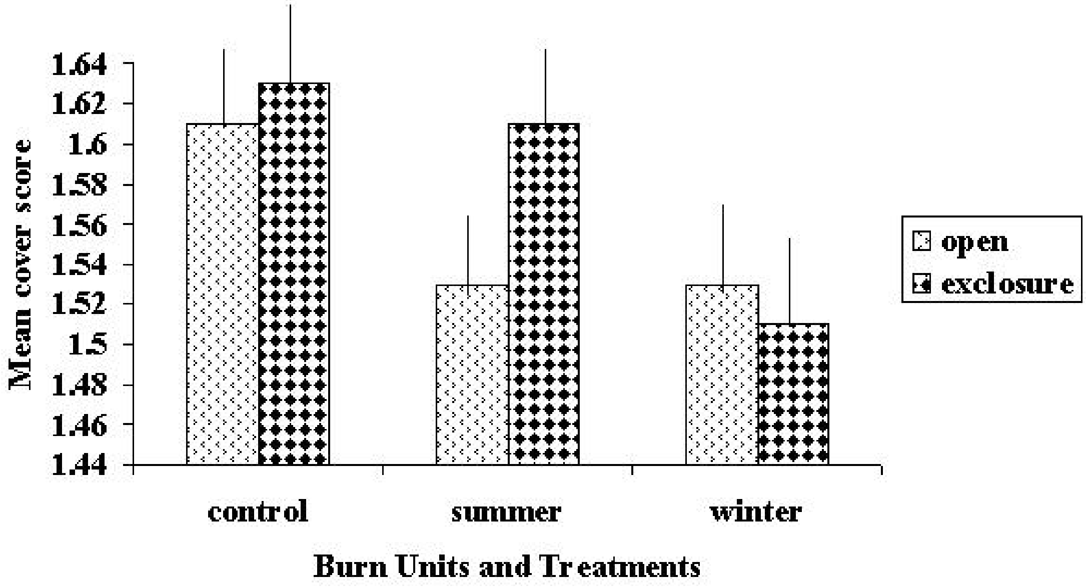


Figure 7.4. Mean cover score for all species at one year after treatment + S.E.



Cover score: 1 = <1%; 2 = 1-5%; 3 = 5-10%; 4 = 10-25%; 5 = 25-50%; 6 = > 50%

Figure 7.5. Species richness after one year (Iris Block only) + S.E.

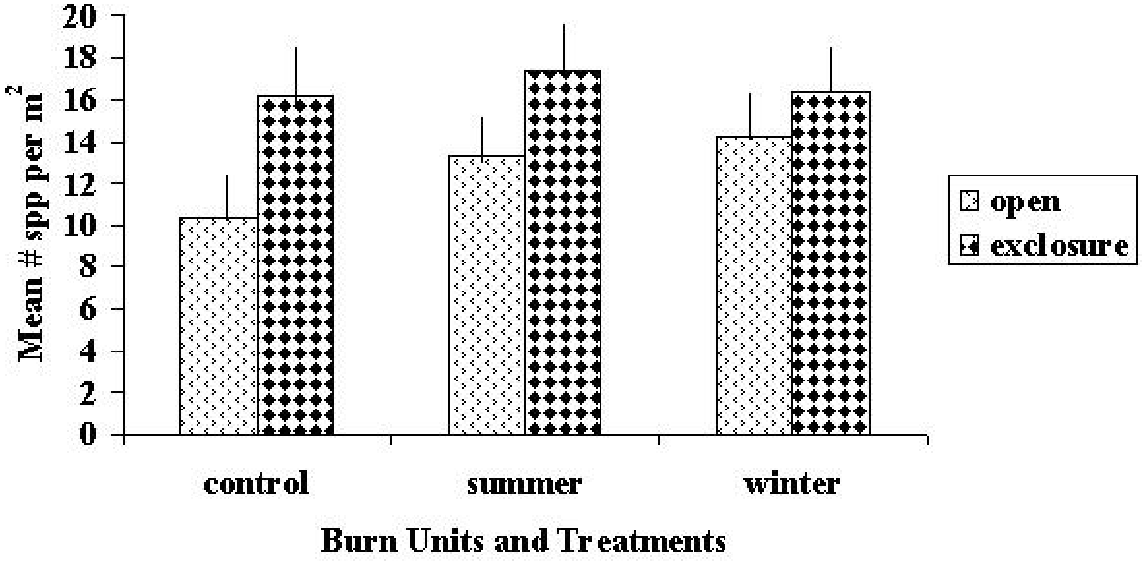
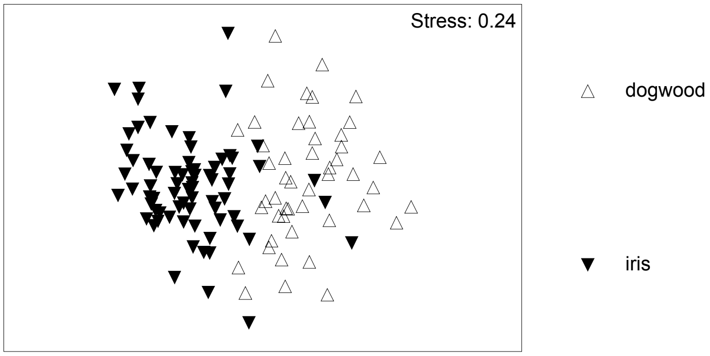
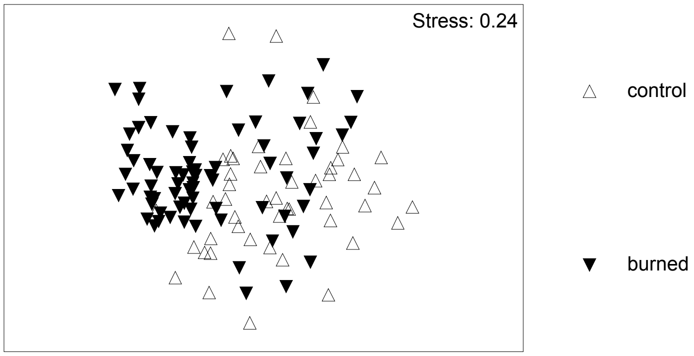


Figure 7.6. MDS 2-D ordination plots based on plant species cover in key deer herbivory plots in Dogwood and Iris blocks. Sample similarity was measured based on Bray-Curtis Coefficients on non-transformed data. Points were labeled with A) block; B) fire treatment; C) exclosure status.

A.



B.



C.

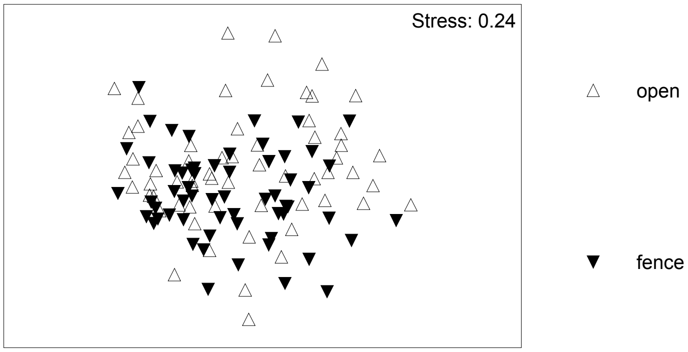


Figure 7.7. MDS 2-D ordination plots based on plant species abundance in key deer herbivory plots in Dogwood and Iris blocks. Sample similarity was measured based on Bray-Curtis Coefficients on non-transformed data. Points were labeled with A) block; B) fire treatment; C) exclosure status.

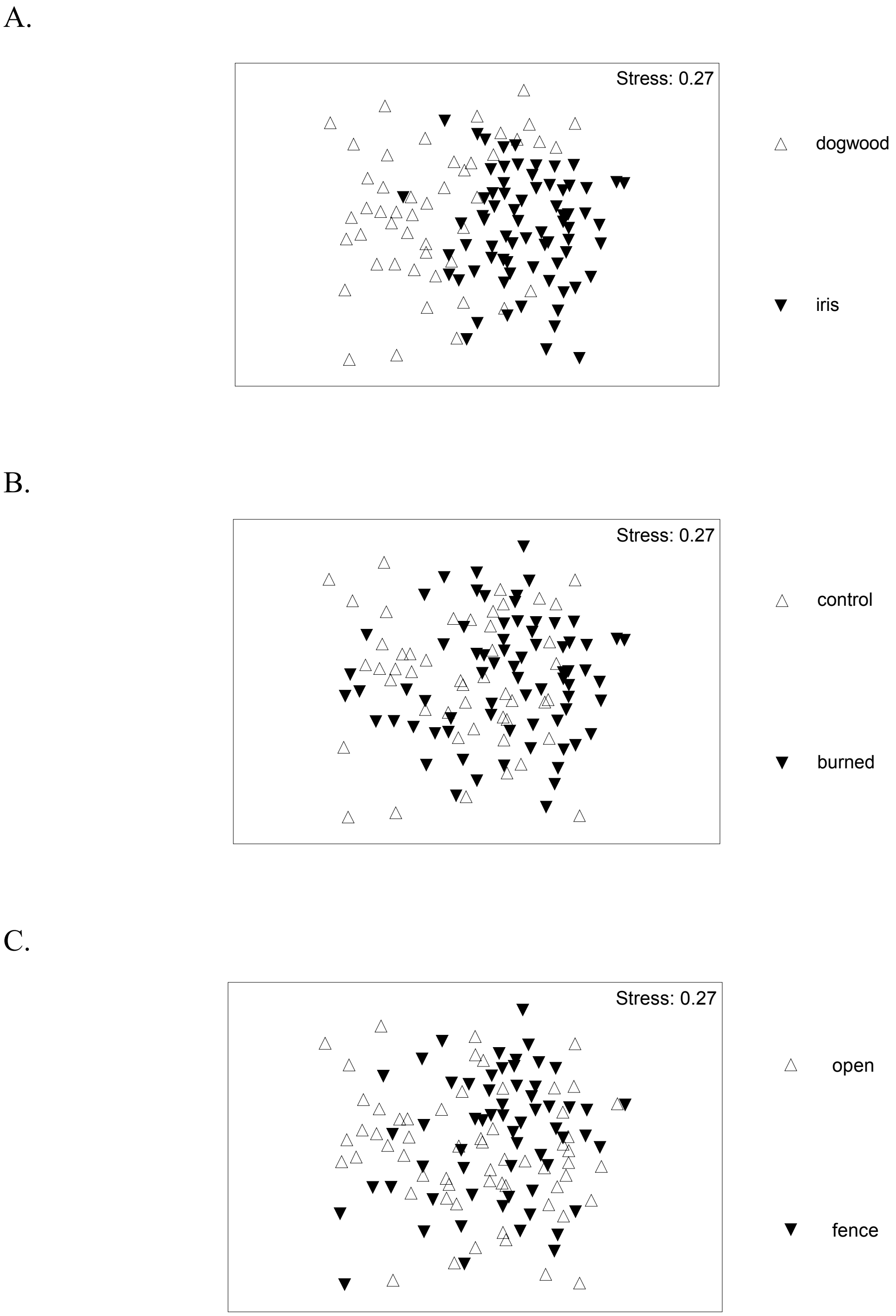
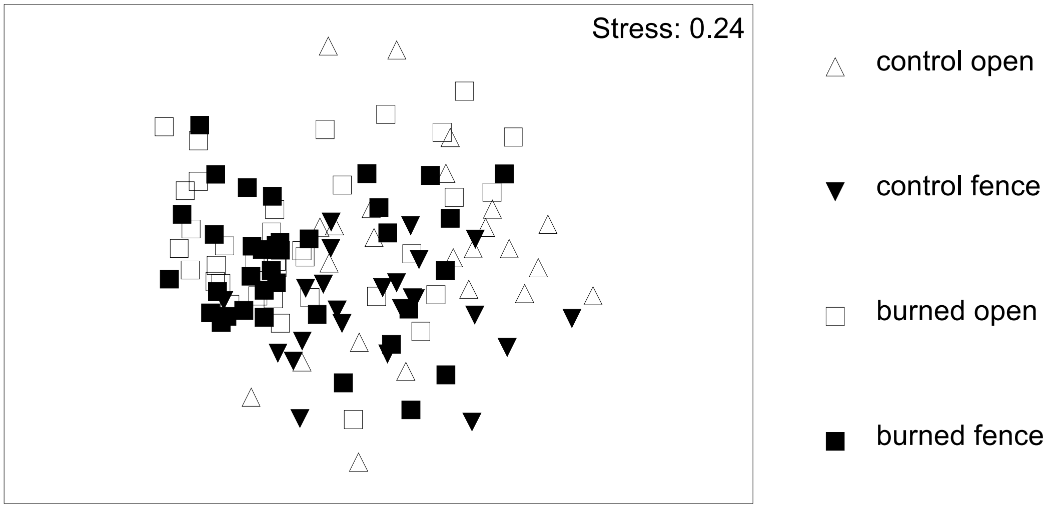
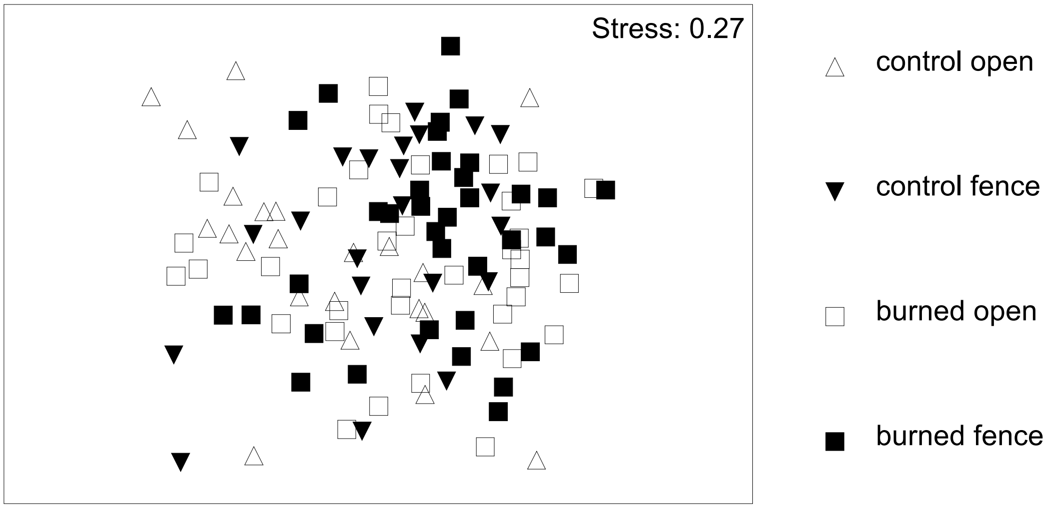


Figure 7.8. MDS 2-D ordination plots with points labeled with fire treatment and exclosure status. Sample similarity was measured based on Bray-Curtis Coefficients on non-transformed data. A) Data based on plant cover; B) Data based on plant abundance.

A.



B.



8. BIOMASS DYNAMICS

Introduction

Through the process of photosynthesis, forests utilize the energy of the sun to fix atmospheric carbon into biomass. Eventually, a fire occurs and some of the stored carbon and potential energy are released back into the environment. The fire cycle is central to the evolution and function of pine forest flora and fauna worldwide; locally, its manipulation is the primary tool available for managing the structure and composition of south Florida pine rocklands, including those of the Florida Keys (Alexander 1967, Alexander and Dickson 1972, Snyder et al. 1990, Slocum et al. 2003). The development of strategies to manipulate the fire cycle effectively requires a comprehensive understanding of fuel dynamics and the effect of fuels on fire behavior.

Immediately after a fire, available fuels are often insufficient to carry a second fire. The rate of accumulation of fuel loads in forest stands is thus an important determinant of fire frequency and severity (Davis and Cooper 1963; Paatalo 1998; Cochrane et al. 1999). As such, fire return interval varies among forest types, based on site characteristics that affect productivity (Beaty and Taylor 2001; Stephens 2001). Information regarding the nature and distribution of fuels in relation to fire intervals has been used to project how rapidly fires will spread, their intensity, and ultimately their ecological effects (Rothermel 1972; Kauffman et al. 1994; Paatalo 1998; Baeza et al. 2002).

Season of fire is also an important component in the fire regime of south Florida pine forests, as it may influence fuel consumption, fire intensity, and forest development during the post-burn period (Kaufman and Martin 1989; Robbins and Myers 1992; Spier and Snyder 1998; Negrón-Ortiz and Gorchov 2000; Ryan 2002; Sparks et al. 2002). Fire intensity is usually, though not invariably, higher in dry- than wet-season fires (Menges and Deyrup 2001). In addition to its effects on existing vegetation, a very intense fire may hinder recovery of the plant community (Spier and Snyder 1998), thereby constraining the development of fuel loads in the post-burn period. The effect of fire seasonality on development of fuel loads in Florida Keys pine forests has not been studied directly; models developed more generally for the southeastern U.S are applied, but may not fit the peculiarities of Keys' climatic and edaphic conditions well.

A mean fire return interval may be defined for any location, but spatial and temporal variability around this central tendency is typically large, with significant implications for fire behavior. A long interval between fires usually facilitates fuel accumulation, mostly in the form of fine and dead biomass, resulting in intense fire (Rothermel 1972; Van Wilgen 1982; Hobbs and Gimingham 1984; Reggan et al. 1988; Bradstock et al. 1998). On the other hand, a short fire interval drastically reduces the biomass (Van Wilgen and Kruger 1981). In general, plant fuel loads increase with time after burning and then become constant (Chaffey and Grant 2000; Baeza et al. 2002; McCaw et al. 2002). Researchers in several ecosystems have reported that after some minimum critical period, structural changes in vegetation and hence in fuel distribution occur, and that these changes are particularly relevant for fire behavior in prescribed burning management (Baeza et al. 2002). The detection of this critical period in the fuel cycle is important, especially where the main strategy of prescribed burning is fuel reduction (Rothermel 1972; Koehler 1993). Fuel reduction has been an important consideration in the management of Florida Keys pine forests, where prescribed burning has been applied sporadically since the creation of National Key Deer Wildlife Refugee in 1957 (Bergh and Wisby 1996). From 1985 to 1992, the primary objective of burns conducted in the Refuge was to reduce fuels and to prevent intense wildfires that would cause excessive pine mortality. After this active period of management, the prescribed fire program was inactive for about 6 years, resulting in forests of various stages of development following fire.

In this study, our objectives were: i) to estimate understory and overstory biomass in pine rockland forests, ii) to examine the effects of season on understory fuel consumption, and iii) to examine the dynamics of understory fuel load accumulation after fire.

Methods

The study design and site descriptions have been presented thoroughly in earlier chapters. We were able to estimate the major aboveground biomass components (described below) at least once in 18 stands, which varied initially from 8 to more than 30 years in the time since their most recent fire (Table 8.1). Eleven of the 18 stands were burned under prescription during the study, as described previously. In these, we were able to examine the amount and types of fuel consumed and, in all except those burned during the final year of the project, the early stages of post-fire fuel accumulation.

Biomass estimation

We estimated biomass within the following broad structural categories: i) ground layer, ii) shrubs, iii) palms iv) pines, and v) hardwood trees. Methods of biomass estimation differed slightly among the groups, and are described below.

Ground layer**:** Ground layer fuels (consisting of litter and vegetation <1 m in height) were measured in all plots, including Control plots, prior to the burns in each block. In the 11 plots subjected to experimental burning, ground fuels were measured again within one month following the fire, allowing estimation of fuel consumption. Finally ground fuels were collected in all 18 plots at the end of the project (2000), enabling us to directly assess the accumulation of fuels over the course of time. We located 40 1-m^2 plots within each 1 ha burn unit. Ground fuel plots were centered on the 20 shrub plots, whose location had been determined by stratified random selection; two ground fuel plots were distributed at 4.5 m distance from the shrub plot center in opposite directions. One previously delineated 0.25-m^2 sub-plot per plot was harvested during each sampling event.

Plot harvest was accomplished by (1) excising stems at ground level and collecting all standing plant material less than 1 m in total height, including palms, and (2) collection of all undecomposed litter and coarse woody debris <2.5 cm in diameter. Samples were separated into live and dead fuels, and further into the following categories: forbs, ferns, graminoids (including grasses, sedges and grass-like forbs), woody plants, palms, litter <0.5 cm, and litter ≥0.5 cm but < 2.5 cm in diameter. Separated samples were oven dried for 48 hrs at 70° C and weighed.

Shrub fuels***:*** Shrub biomass was estimated by applying regression models to plant dimensional data. Within each plot, plant dimensions of woody stems >1 m height and <5 cm diameter rooted within 20 circular, 4-meter radius plots were measured. Measurements differed for shrub-like and tree-like species. For shrub-like species, we treated clumping stems as a single individual, and measured height and two crown diameters (the longest, and its perpendicular), whereas for tree-like plants we measured total height and DBH. Regression models based on these measurements were developed from a sample of about 15 stems of each of the major shrub species, collected outside the study plots on Big Pine Key (Sah et al. 2004).

Palms**:** Total palm biomass was calculated as the summation of trunk and leaf biomass, which were estimated separately from palm dimensional data, as described below. Because height is often a good predictor of palm biomass (e.g., Frangi and Lugo, 1985), we measured total height and apical height (height to the apical meristem) of each individual belonging to 4 palm species (*Coccothrinax argentata, Sabal palmetto, Serenoa repens* and *Thrinax morrisii*) within the shrub plots described above, regardless of stem diameter. For biomass estimation purposes, we categorized palm individuals into ten size-classes based on apical height.

To estimate the biomass of palm trunks, it was necessary to estimate the bole volume and wood specific density for each tree sampled. Assuming a cylindrical form for a palm trunk, we calculated volume ($v = \pi(d/2)^2h$) of each individual tree, where h was the measured apical height, and d was the mean basal diameter (measured at 10 cm above palm-butt) obtained from 3 to 5 randomly selected individuals in each of the ten height classes. Wood specific density (SD) was calculated for a pie-slice or cylinder excised from the trunk of six trees, three each of *Coccothrinax argentata* and *Thrinax morrisii*, using the oven dry method (Barajas-Morales 1987). Wood sections were dried at 70°C to a constant weight. Their volumes were determined by the mass of water displaced when the sample was completely submerged in a beaker of water placed on a digital balance of 2000g capacity. The increase in weight in grams was equivalent to the volume of the palm slice in cubic centimeters. Specific density was calculated as the ratio of the oven-dry mass divided by the volume of the wood samples. Trunk biomass was then calculated for each tree as the product of wood specific density and bole volume.

Palm leaf (blade + petiole) biomass was estimated according to methods described in Cooley (2004). She found that height was a poor predictor of palm leaf biomass, and developed equations based on crown area and leaf number, which were parameters that were not included in the shrub sampling protocols. Therefore, in order to obtain estimates of live and dead leaf biomass for *C. argentata* and *T. morrisii*, we multiplied the number of censused individuals in each palm size-class by the mean live and dead standing leaf biomass of an average individual within the class. We determined the mean leaf biomass in each size class by applying Cooley's best regression equations to ten randomly chosen individuals per class at six sites (Cooley 2004).

Pines: Pine sapling (height >1 m and DBH <5 cm) biomass was estimated by using a regression developed locally for *Pinus elliottii* var. *densa* stems of this size Independent variables tested were height and DBH. 13 individuals were measured, collected, and weighed from several locations within the study area. The best regression model was applied to dimensional data collected from the shrub plots, in which height and DBH of pine saplings were measured.

Pine tree biomass was estimated using equations developed by Gholz and Fisher (1982) for various tree components. The regression models were applied to the tree DBH data described in Chapter 3. Using DBH as a predictor, we calculated the dry weight of stem bark and wood, live and dead branches, and total foliage, and summed them to derive a total aboveground biomass of pine trees.

Hardwood trees**:** Biomass of hardwood trees (DBH >5 cm) was estimated using a generalized equation developed by Brown et al. (1989) for dry tropical forests. The regression equation was applied to the hardwood tree DBH data, as described above for pines.

We defined available fine fuels, or fuel loads, to include: (1) total ground layer biomass (live and dead stems <1 m height + litter and coarse woody debris in the forest floor), (2) shrub leaves and twigs < ¼ inch in diameter, (3) palm leaves and petioles, and (4) pine sapling needles and twigs.

Analytical methods

We used one-way analysis of variance (one-way ANOVA) to assess the significance of differences among blocks in terms of total biomass and available fuel loads. Considering the pre-burn data, we applied a space-for-time substitution at the plot level to assess the association (Pearson's *r*) between several biomass components and time since fire. One-way ANOVA was also used to evaluate differences among burned plots in terms of ground fuel consumption. Relationships of ground fuel consumption with preburn ground fuel and time since last fire were examined by fitting a linear regression equation to the plot-level data using standard least squares techniques. Only two blocks, Orchid and Poisonwood, were burned in both wet and dry seasons. Two-way analysis of variance (two-way ANOVA) was used to examine the effects of Block and Season on ground fuel consumption calculated for 20 shrub subplots in the four burned plots in these blocks.

Stepwise-multiple regression with forward selection was used to examine the effects of local variation in preburn fuel and fire intensity on ground fuel consumption. Independent variables were preburn ground fuel, scorch percentage and char height, calculated for 20 shrub subplots in the two Orchid and two Poisonwood plots. At each step, the entered predictor was evaluated for significance on the basis of the 't' statistic ($p < 0.05$) and by its contribution to the total variation explained in the model. Finally, only the significant predictors were retained in the model. For each model, the assumptions of least-square regression were tested by examining the normality of standard residuals. Statistical analyses were carried out using STATISTICA version 6.0 (StatSoft, Inc. 2001).

Results and Discussion

Pre-burn biomass and fuel loading

Mean total aboveground biomass in Big Pine Key pine forests (including ground fuels) averaged 76.2 Mg ha^{-1} (Figure 8.1). Among-block variation was significant ($F_{5,\ 12}$ = 17.1, p <0.001), with pine forests in the Poisonwood block exceeding other local forests in biomass (105.4 Mg ha^{-1}). Pine trees constituted more than half (53.3%) of the biomass, and palms, the dominant understory contributor, accounted for 18.7%. Broad-leaved shrubs and ground layer fuels constituted 4.2% and 21.8% of total biomass. Hardwood trees, though well-distributed throughout the forests, comprised a relatively small percentage (1.4%) of stand biomass. Biomass in the relatively well-developed Poisonwood block was comparable to Snyder's (1986) estimates for Long Pine Key in Everglades National Park (75-90 Mg/ha in standing vegetation, 10-17 Mg/ha in litter). However, biomass in both of these south Florida pine forests is far less than in 25 year-old north Florida slash pine plantations, in which total aboveground biomass values >150 Mg/ha are attained (Gholz and Fisher 1982).

Total stand biomass was uncorrelated with time since fire (r = +0.12, p>.05), but the biomass of palms, broad-leaved shrubs and hardwood trees increased with fire-free period (r = +0.62, +0.59, and +0.75, respectively; p<.05 for all three). The correlation analyses imply, albeit indirectly, that palms, shrubs, and hardwood trees are dynamic structural components, i.e., they are relatively sensitive to non-stand replacing fires, but recover quickly thereafter. This is surely true for shrubs and hardwoods, which resprout prolifically after fire, but palms grow very slowly (Cooley 2004) and do not resprout; thus, the strong relationship between total biomass and time since last fire in this group may result from some covariant factor, perhaps related to long-term fire history. Direct information regarding mortality and postfire recovery in these groups is presented in Chapters 3-5 of this report.

Available fine fuels in lower Keys pine forests constituted 30% of the total aboveground biomass (Figure 8.2), a proportion in substantial agreement with Gholz and Fisher (1982). On Big Pine Key, about three-fourth of available fine fuels prior to the experimental fires were in the ground layer. Palm leaves were 21.5% of available fine fuels, and broad-leaved shrubs comprised another 5.3%. Pine saplings contributed very little to the total fuel loads (<1%). One fine fuel category not included in Figure 8.2 is attached pine needles in the forest canopy, because little is directly consumed in most prescribed fires. This fuel component comprises only 5-10% (among-stand range: 1.4-2.5 Mg/ha^{-1}) of the available fine fuel.

Among-block differences in fuel loads were significant ($F_{5,\ 12}$ = 18.0, p <0.001), with the Locustberry block (14.3 Mg ha^{-1}) containing only about half the fuel loads present in the Poisonwood and Buttonwood blocks (28.8 Mg ha^{-1} each) (Figure 8.2). Fuel loads generally increased from the most recently burned stand (Iris-8) to the longest unburned community (Buttonwood). However, the correlation of available fuels with time since fire was non-significant ($r = 0.20$), primarily because of anomalously low fuel loads in stands burned 14 years before sampling in the Locustberry block and one portion of the Iris block. We suspect that these two areas may have burned intensely during their most recent fire, perhaps resulting in high mortality of palms, broadleaved shrubs, and mature pine trees, and therefore slow and patchy accumulation of ground fuels compared to neighboring forests.

Ground fuel consumption and recovery

During the study period, a total of 11 plots, including at least one plot per block, were burned (Table 8.1). Eight plots were burned in the summer (June-August), and three plots, one each in the Iris, Orchid and Poisonwood blocks, were burned in early winter (December). Based on post-fire and pre-fire sample weights, prescribed fire consumed an average of 56.8% of the ground fuel across all plots. Among-plot differences in the rate of fuel consumption were significant (Figure 8.3). However, such differences were not related to time since last fire ($n = 10$, $R^2 = 0.066$, $p = 0.475$). Percentage ground fuel consumption tended to be higher where pre-burn ground fuels were most abundant (Figure 8.4), but the relationship was not significant at the plot level ($n = 10$, $R^2 = 0.175$, $p = 0.228$). Likewise, at the plot level, percentage consumption of ground fuels was not related to crown scorch percentage or char height, two commonly used surrogates of fire intensity (e.g., Menges & Deyrup 2001).

In two blocks, Orchid and Poisonwood, plots were burned in two different seasons. Two-way analysis of variance indicated that the percentage of ground fuels consumed was significantly affected by both Block ($F=6.98$, $p=.010$) and Season ($F=6.62$, $p=.012$), but the Block x Season interaction was non-significant. As illustrated in Figure 8.5, consumption percentages were slightly higher in summer than winter fires, and in the Orchid than the Poisonwood Block.

The analyses described above focus at the plot level, where patterns evident at a smaller scale may be averaged out, potentially obscuring important relationships. To examine ground fuel consumption at a finer scale, we used multiple regression to assess how ground fuel consumption calculated individually for each of the twenty shrub plots per stand was affected by (1) the initial, pre-burn quantities of ground fuel present in each plot, and (2) plot estimates for char height and canopy scorch (see Chapter 2 for methods of estimation). The regression analyses were applied to combined data from the four burned plots in the Orchid and Poisonwood blocks. These data represent a broad range of pine rockland conditions, in that they included both summer and winter fires, carried out in relatively open (Orchid) and closed (Poisonwood) understory conditions. Among the independent variables, pre-burn ground fuels ($p=.0004$) and char height ($p=.006$) were significantly related to fuel consumption, but crown scorch was unrelated ($p=.10$). The equation for the 2-factor model was:

$$\% \text{ consumed} = 11.80 + 0.15 * (\text{pre-burn fuels, g/m}^2) + 6.7 * (\text{char height, m}),\ R^2 = .306$$

Thus, percent consumption increased with ground fuel mass and with fire intensity. Figure 8.6**,** which is based on a distance weighted least squares function applied to the raw data, clarifies the influence of both factors on fuel consumption. The function suggests that the effect of both ground fuel mass and fire intensity are greatest when the other factor is low.

The assessment of post-fire ground fuel dynamics requires a space-for-time substitution, in that a temporal sequence is inferred from ground fuel biomass estimates for plots burned in different years, where the estimates were made during a common period, i.e., at the conclusion of the project. Figure 8.7 presents ground fuel loads for seven plots that varied in time since prescribed fire from 1-3 years. While the distribution suggests some buildup of ground fuels during this early stage in recovery, the accumulation rate can be calculated directly for each plot. On average, estimated accumulation rate for ground fuels at these sites was 0.53 Mg/ha/year (SE = 0.54). These estimates, which range from 1.20 Mg/ha/yr in the 3 year-old PS plot to -0.26 Mg/ha/yr in 2 year-old IS, indicate very high levels of site variation. Negative values may result from relatively high decomposition rates (Snyder 1986), or they may simply reflect random microsite variation among adjacent quadrats, to which the sequential harvest method we used is especially sensitive (see Methods). In the slash pine plantations studied by Gholz and Fisher (1982), ground fuels accumulated at a constant rate slightly exceeding 1 Mg/ha/yr through age 35.

Shrub layer fuel dynamics

Our sampling protocols did not allow a direct estimation of the consumption of shrub fuels. More specifically, we could not simply subtract pre-burn from post-burn standing crop, as we had done for ground fuels, because of the complicating effects of re-growth during the one-year period prior to initial post-burn sampling. However, for plots burned early in the project (PS, PW, OS, OW, DS, IS, and IW), it was possible to estimate shrub fuel accumulation during Year 2 and in some cases Year 3 after fire. Figure 8.8 displays the decline in fine fuels associated with palms and broad-leaved shrubs from the pre-fire condition to Year 1 post-fire, and then the recovery through Year 3 across all available sites. The initial decline in palm fuels was less steep than for broad-leaved shrubs (70% and 88%, respectively), probably because surviving palms leafed out rapidly from pre-fire meristems, and were immediately incorporated in the shrub layer. In contrast, broad-leaved shrubs were usually top-killed; when they survived, they generally re-sprouted from the base of the original shoots, and did not reach 1 meter height by the end of Year 1. After Year 1, both groups recovered rapidly, with palms and shrubs reaching 70% and 47%, respectively, of their pre-fire fine fuel biomass by Year 3. Cooley (2004) demonstrated that attached palm fine fuels tended to reach an asymptote well within a decade after fire in the Keys, while fine fuels associated with broad-leaved shrubs may take slightly longer to reach a maximum (Sah et al. 2004).

Conclusion

Fine fuels represent a very substantial proportion of the total aboveground biomass in Florida Keys pine forests. For this reason, fuel regulation via prescribed fire may be particular critical if catastrophic, stand-replacing fires are to be avoided. The presence in the landscape of a rich pine rockland understory flora provides some evidence that this sort of regulation took place naturally, or perhaps with the involvement of aboriginal inhabitants. Today, the constraints imposed by encroaching residential users make lightning a less viable source of ignition, and place a heavy burden on resource managers to design fuel management techniques that allow humans to coexist with a healthy pine rockland ecosystem.

In conjunction with earlier work reported by Cooley (2004) and Sah et al. (2004), the data reported here form an initial basis for developing such techniques. Our data provide evidence of rates of accumulation in several major fuel components, as well as standing stocks of fuels in forests of variable history. The data also suggest that fuel accumulation and season of burn do affect fire behavior, and perhaps vegetation response. Most of all, the results suggest that we need to become more analytical about the relationships between fire weather, fuel amounts, forest structure, and fuel conditions, and go beyond simplifications that have been depended on till now.

Table 8.1: Study sites with burning date and sampling years. * Unburned, + Immediate postburn, @ Annual postburn

Site	Burning Treatment	Plot ID	Year when plots were last burn	Date of experimental burn	Year of sampling							
					Shrub, Palms & Saplings				Ground layer biomass			
					1998	1999	2000	2001	1998	1999	2000	2001
Buttonwood	Control	BC	Before 1970	Not burned			*				*	
	Summer	BS	Before 1970	7/8/2001			*				* +	
	Winter	BW	Before 1970	7/8/2001			*				* +	
Dogwood	Control	DC	1988	Not burned		*	*	*		*		*
	Summer	DS	1988	6/22/1999		*	@	@		* +		@
	Winter	DW	1988	Not burned		*				*		
Iris	Control	IC	1991	Not burned		*	*	*		*		*
	Summer	IS	1991	7/14/1999		*	@	@		* +		@
	Winter	IW	1985	12/12/2000		*		@		*		@
Locust Berry	Control	LC	1986	Not burned			*			*		
	Summer	LS	1986	7/19/2001			*			* +		
	Winter	LW	1986	7/19/2001			*			* +		
Orchid	Control	OC	1990	Not burned	*	*	*	*	*			*
	Summer	OS	1990	8/16/1998	*	@	@	@	* +			@
	Winter	OW	1990	12/15/1998	*	@	@	@	* +			@
Poisonwood	Control	PC	1986	Not burned	*	*	*	*	*			*
	Summer	PS	1986	8/17/1998	*	@	@	@	* +			@
	Winter	PW	1986	12/15/1998	*	@	@	@	* +			@

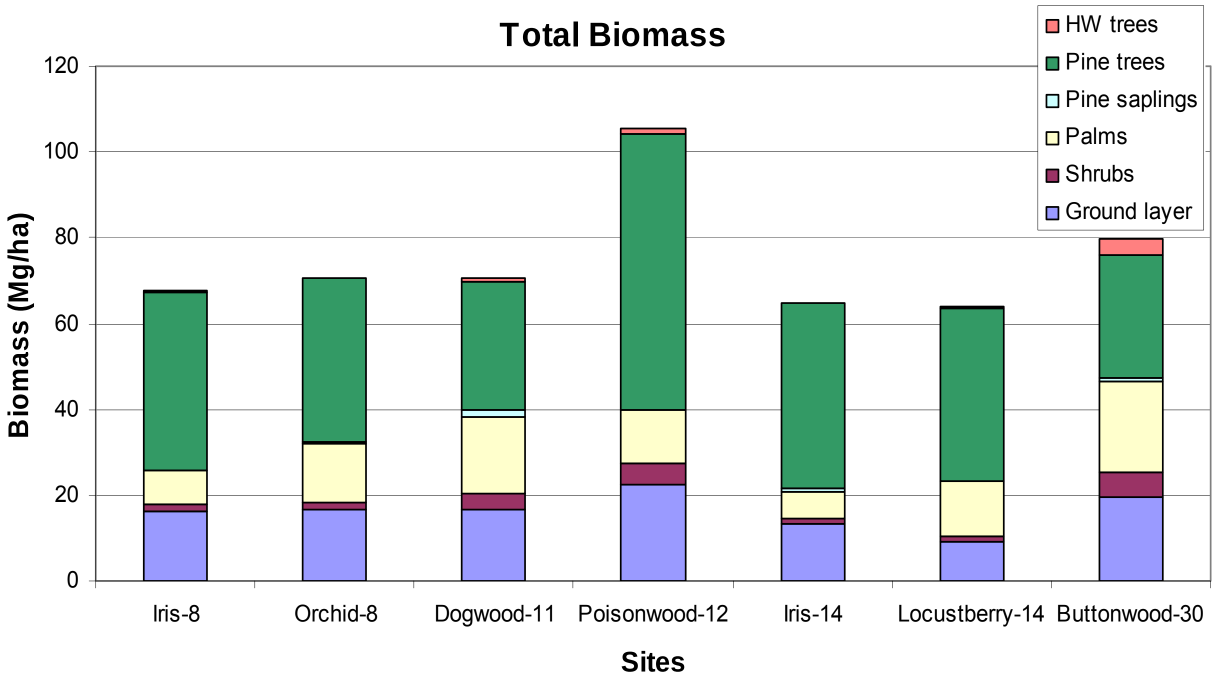


Figure 8.1. Total biomass in six blocks of Big Pine Key pine forests sampled prior to experimental fire. The numbers associated with site name represent time (years) since last fire. Pine forests in the Iris Block consist of two zones, one was burned 8 years and the other 14 years before recent sampling.

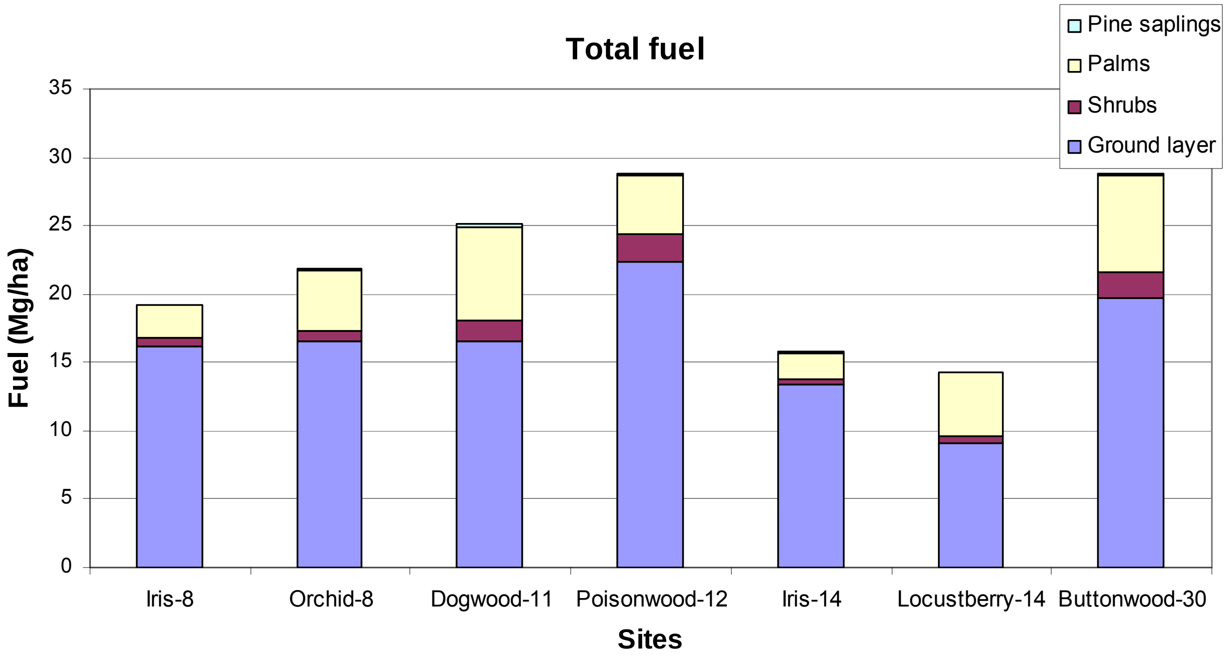


Figure 8.2. Total available fuels in six blocks of Big Pine Key pine forests sampled prior to experimental fire. The numbers associated with site name represent time (years) since last fire. Pine forests in the Iris Block consist of two zones, one was burned 8 years and the other 14 years before recent sampling.

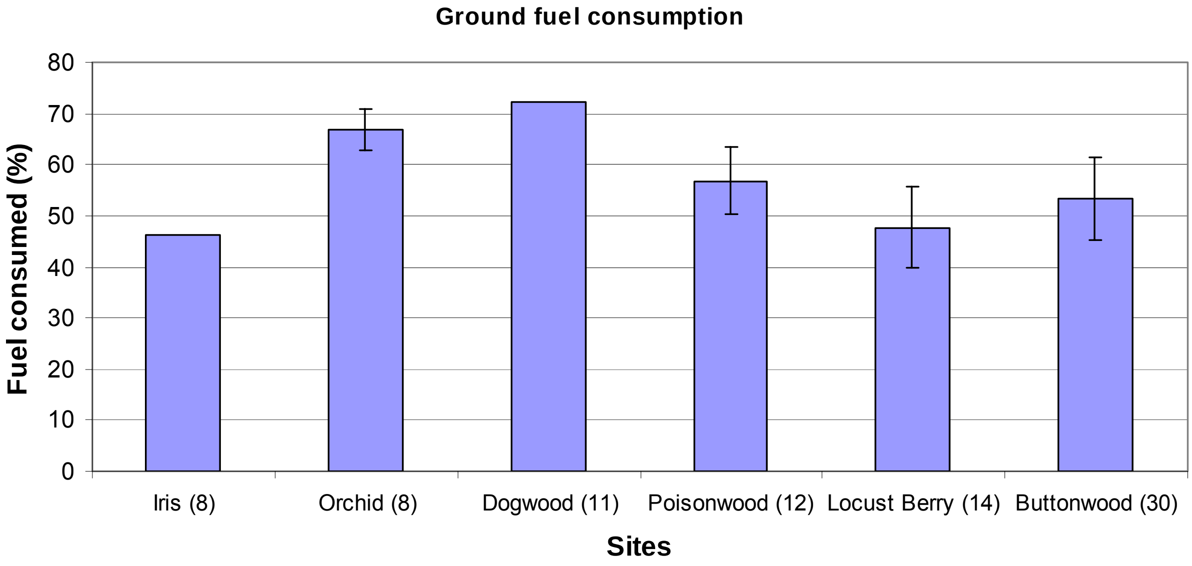


Figure 8.3. Mean percentage of ground fuel consumption during experimental burnings in six blocks of Big Pine Key pine forests. The numbers in parenthesis are time (years) since last fire.

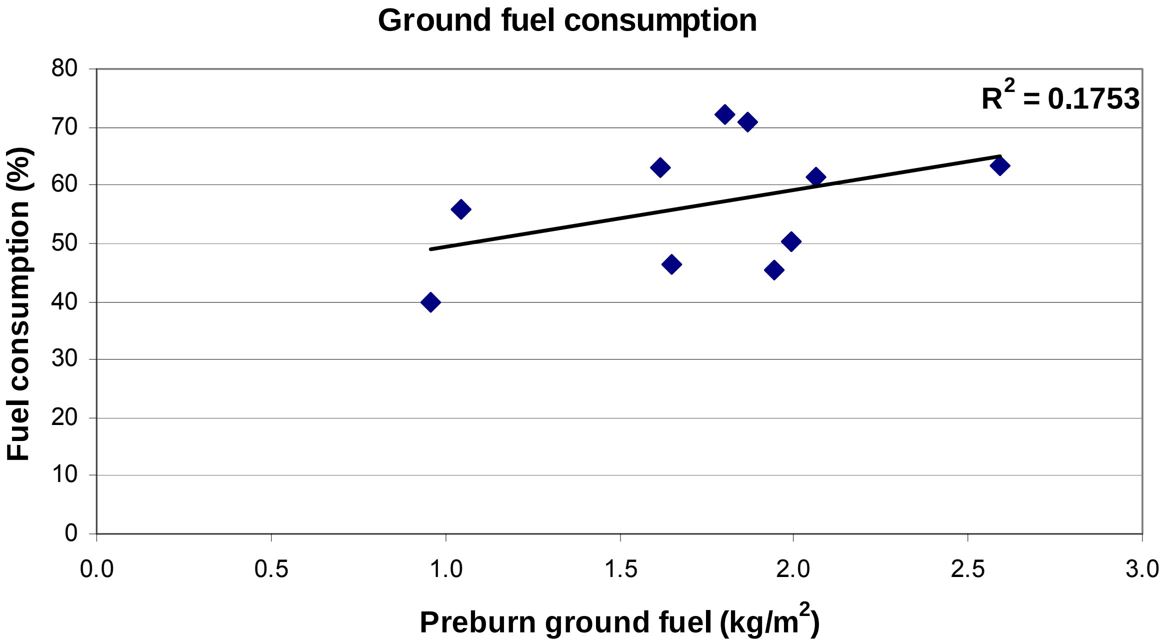


Figure 8.4. Mean percentage of ground fuel consumption during experimental burnings in relation to pre-burn ground fuel present in Big Pine Key pine forests.

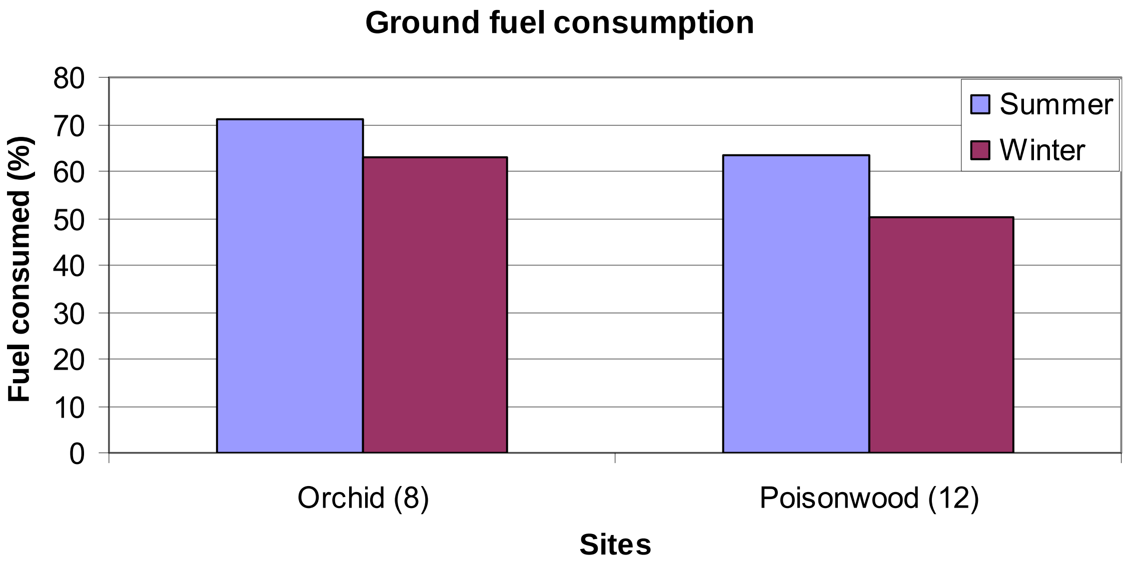


Figure 8.5. Mean percentage of ground fuel consumption in the pine forests of Orchid and Poisonwood blocks, burned in two different seasons, summer and winter. The numbers in parenthesis are time (years) since last fire.

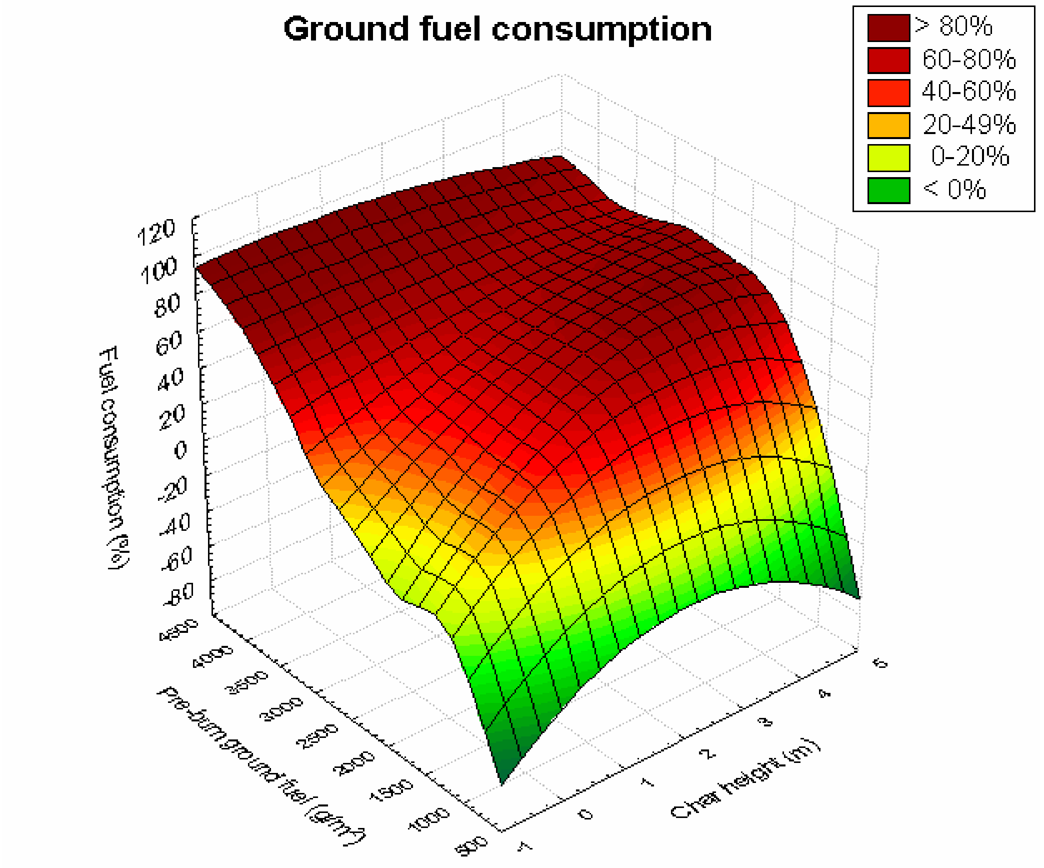


Figure 8.6. Mean percentage of ground fuel consumption in relation to pre-burn ground fuel and char height during the experimental burning in the pine forests of Big Pine Key.

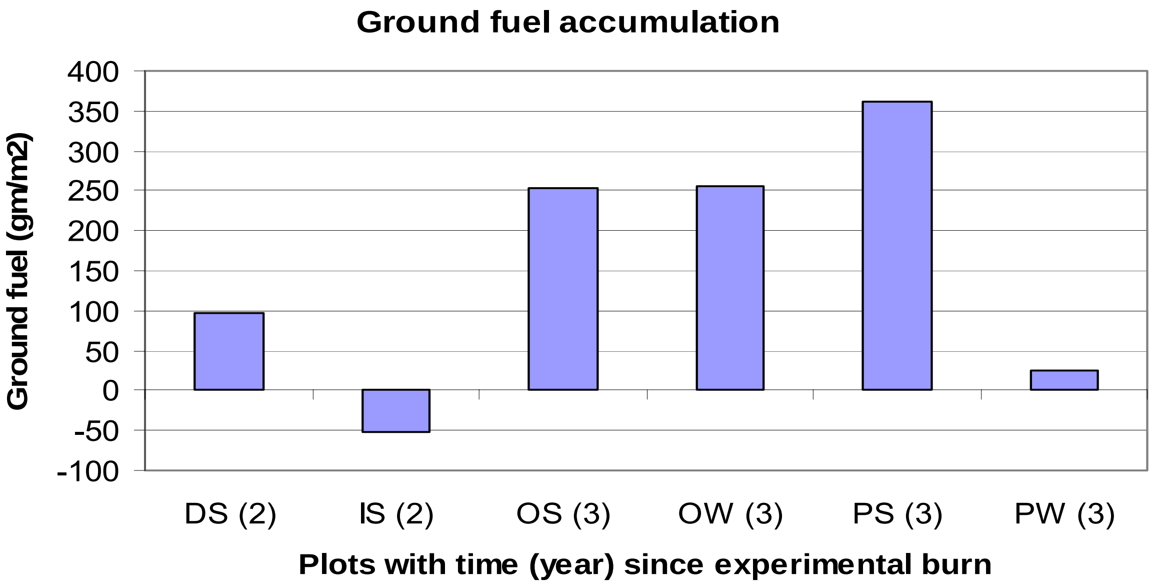


Figure 8.7. Ground fuel accumulation after experimental burn in the pine forests of Big Pine Key. Numbers in parenthesis are the time since experimental burn.

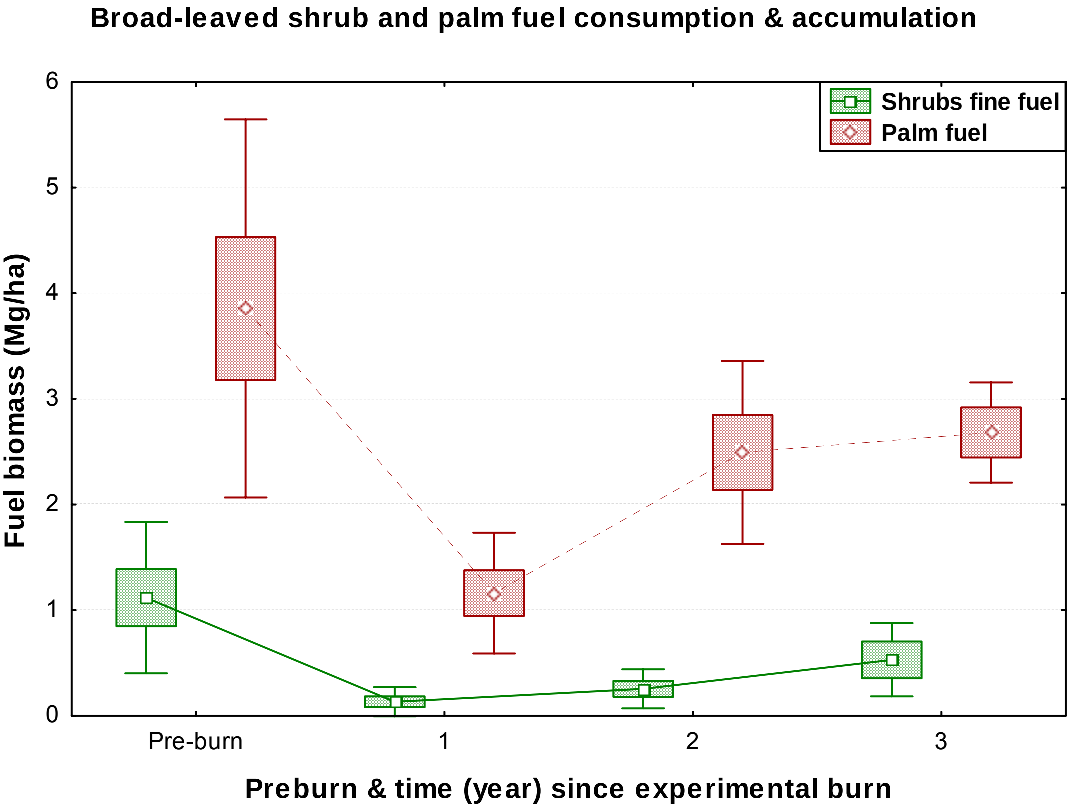


Figure 8.8. Shrub fine fuel and palm fuel prior to and 1-3 years after experimental burn in the pine forests of Big Pine Key.

9. CONCLUSIONS

The task of prescribed burning is not easy—witness the difficulty we had in getting a series of simple experimental burns done on schedule. External factors delayed the experimental treatments and did not allow us to collect postburn data on one third of the burns.

In the three blocks in which paired summer and winter burns were successfully carried out, the summer burns had more intense fires as judged by our indicators of fire intensity (crown scorch, height of bark char, fire temperature, fuel consumption). Under other circumstances that might not always be the case.

Because the summer burns were more intense than the winter burns, it is not possible to say whether differences in vegetation response between summer and winter burns are due to season or to fire intensity.

The mortality of South Florida slash pine trees was greater after the summer burn than the winter burn in each block, but there was rarely such a consistent pattern seen for other vegetation responses. One need only look at the recovery of stem numbers of common shrub species after burning: *Metopium* showed less recovery after summer burns in Orchid and Iris and after the winter burn in Poisonwood. *Pisonia*, while it also had less recovery after the summer burn in Iris and the winter burn in Poisonwood, showed more recovery after the summer burn at Orchid.

The same species may respond differently to the same burns depending on the growth stage of the plant. Adult palms were generally more likely to succumb to summer than winter burns, and *Coccothrina*x was much more likely to die than *Thrinax*. However, the recovery of small palms was greater after summer burns than winter burns.

Higher intensity fires may be beneficial for some plant responses such as germination of *Jacquemontia pentantha* seeds.

Fine fuels represent a substantial proportion of the total aboveground biomass in Lower Keys pine rocklands.

During the course of this study a series of permanently marked vegetation plots were established. There are now 18 permanently marked 1.0-ha plots on Big Pine Key in which all trees with DBH $\geq$ 5 cm were tagged and measured. Plots should be periodically re-measured to follow growth, mortality, and recruitment into the tree stratum. More than 11,400 trees were tagged. Similarly, within each of the 18 plots there are 20 marked shrub plots in which woody plants and palms $\geq$ 1 m tall were tallied, and 80 individually marked permanent herb quadrats in which all the species were enumerated. These plots will be immensely valuable to document change, whether due to prescribed fire, wildfire, hurricane, sea level rise, or some other unforeseen disturbance.

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12. APPENDICES

Appendix 1. GPS coordinates for corners of 18 experimental plots. Projection=geographic, Datum=WGS-84, Units=Decimal Degrees.

BLOCK	PLOT	CORNER	XCOORD	YCOORD	GPSDATE	GPSTIME
Buttonwood	BC	SW	-81.374275	24.688400	5/9/2000	11:20:02 AM
Buttonwood	BC	SE	-81.373774	24.688390	5/9/2000	12:15:25 PM
Buttonwood	BC	NW	-81.374170	24.690219	5/9/2000	11:10:45 AM
Buttonwood	BC	NE	-81.373674	24.690202	5/9/2000	11:53:47 AM
Buttonwood	BS	SW	-81.373314	24.688323	5/9/2000	12:20:17 PM
Buttonwood	BS	SE	-81.372809	24.688335	5/9/2000	12:24:27 PM
Buttonwood	BS	NW	-81.373322	24.690143	5/9/2000	11:44:22 AM
Buttonwood	BS	NE	-81.372825	24.690140	5/9/2000	12:39:31 PM
Buttonwood	BW	SW	-81.372412	24.688730	5/9/2000	12:32:27 PM
Buttonwood	BW	SE	-81.371934	24.688788	5/9/2000	02:09:34 PM
Buttonwood	BW	NW	-81.372638	24.690542	5/9/2000	12:41:31 PM
Buttonwood	BW	NE	-81.372146	24.690583	5/9/2000	02:02:15 PM
Dogwood	DC	SW	-81.376248	24.702965	5/16/2000	03:01:39 PM
Dogwood	DC	SE	-81.375267	24.703045	5/16/2000	02:56:34 PM
Dogwood	DC	NW	-81.376337	24.703868	5/16/2000	03:07:10 PM
Dogwood	DC	NE	-81.375368	24.703944	5/16/2000	02:49:38 PM
Dogwood	DS	SW	-81.376411	24.704210	5/16/2000	03:25:43 PM
Dogwood	DS	SE	-81.375434	24.704305	5/16/2000	03:14:46 PM
Dogwood	DS	NW	-81.376530	24.705116	5/16/2000	03:22:29 PM
Dogwood	DS	NE	-81.375531	24.705195	5/16/2000	03:18:51 PM
Dogwood	DW	SW	-81.377874	24.704241	5/16/2000	03:32:48 PM
Dogwood	DW	SE	-81.376896	24.704327	5/16/2000	03:29:28 PM
Dogwood	DW	NW	-81.377938	24.705134	5/16/2000	03:36:13 PM
Dogwood	DW	NE	-81.376979	24.705210	5/16/2000	03:41:06 PM
Iris	IC	SW	-81.386260	24.713014	5/16/2000	02:07:39 PM
Iris	IC	SE	-81.385289	24.713087	5/16/2000	02:25:19 PM
Iris	IC	NW	-81.386338	24.713908	5/16/2000	02:15:09 PM
Iris	IC	NE	-81.385346	24.713978	5/16/2000	02:20:19 PM
Iris	IS	SW	-81.387693	24.712974	5/16/2000	01:46:06 PM
Iris	IS	SE	-81.386686	24.713072	5/16/2000	02:04:41 PM
Iris	IS	NW	-81.387743	24.713876	5/16/2000	01:53:32 PM
Iris	IS	NE	-81.386749	24.713921	5/16/2000	01:58:42 PM
Iris	IW	SW	-81.390951	24.711854	5/16/2000	01:17:13 PM
Iris	IW	SE	-81.389956	24.711922	5/16/2000	01:24:38 PM
Iris	IW	NW	-81.391027	24.712744	5/16/2000	01:12:04 PM
Iris	IW	NE	-81.390036	24.712820	5/16/2000	01:31:46 PM
Locustberry	LC	SW	-81.379506	24.701753	5/9/2000	09:32:50 AM
Locustberry	LC	SE	-81.378517	24.701767	5/9/2000	09:37:06 AM
Locustberry	LC	NW	-81.379518	24.702656	5/9/2000	09:45:53 AM

BLOCK	PLOT	CORNER	XCOORD	YCOORD	GPSDATE	GPSTIME
Locustberry	LC	NE	-81.378535	24.702669	5/9/2000	09:41:25 AM
Locustberry	LS	SW	-81.379698	24.700574	5/9/2000	09:17:24 AM
Locustberry	LS	SE	-81.378716	24.700622	5/9/2000	09:20:42 AM
Locustberry	LS	NW	-81.379752	24.701471	5/9/2000	09:30:56 AM
Locustberry	LS	NE	-81.378772	24.701521	5/9/2000	09:25:11 AM
Locustberry	LW	SW	-81.379690	24.699302	5/9/2000	08:55:43 AM
Locustberry	LW	SE	-81.378694	24.699326	5/9/2000	09:01:20 AM
Locustberry	LW	NW	-81.379712	24.700205	5/9/2000	09:09:16 AM
Locustberry	LW	NE	-81.378733	24.700233	5/9/2000	09:05:39 AM
Orchid	OC	SW	-81.380276	24.707566	5/9/2000	03:20:12 PM
Orchid	OC	SE	-81.379288	24.707630	5/9/2000	03:15:53 PM
Orchid	OC	NW	-81.380343	24.708464	5/9/2000	03:04:40 PM
Orchid	OC	NE	-81.379354	24.708532	5/9/2000	03:10:05 PM
Orchid	OS	SW	-81.382444	24.710979	5/9/2000	04:20:30 PM
Orchid	OS	SE	-81.381465	24.711095	5/9/2000	04:16:46 PM
Orchid	OS	NW	-81.382561	24.711882	5/9/2000	04:08:16 PM
Orchid	OS	NE	-81.381586	24.711986	5/9/2000	04:12:53 PM
Orchid	OW	SW	-81.380998	24.709664	5/9/2000	02:36:46 PM
Orchid	OW	SE	-81.380015	24.709725	5/9/2000	02:42:07 PM
Orchid	OW	NW	-81.381069	24.710551	5/9/2000	02:52:54 PM
Orchid	OW	NE	-81.380088	24.710622	5/9/2000	02:46:13 PM
Poisonwood	PC	SW	-81.388575	24.701740	5/16/2000	11:10:50 AM
Poisonwood	PC	SE	-81.387801	24.701816	5/16/2000	11:06:13 AM
Poisonwood	PC	NW	-81.388720	24.702863	5/16/2000	11:19:02 AM
Poisonwood	PC	NE	-81.387931	24.702940	5/16/2000	11:24:41 AM
Poisonwood	PS	SW	-81.388352	24.699000	5/16/2000	10:50:31 AM
Poisonwood	PS	SE	-81.387660	24.699358	5/16/2000	10:45:00 AM
Poisonwood	PS	NW	-81.388949	24.699974	5/16/2000	10:57:49 AM
Poisonwood	PS	NE	-81.388273	24.700340	5/16/2000	10:35:46 AM
Poisonwood	PW	SW	-81.389021	24.700742	5/16/2000	10:16:26 AM
Poisonwood	PW	SE	-81.387798	24.700848	5/16/2000	10:03:11 AM
Poisonwood	PW	NW	-81.389095	24.701463	5/16/2000	10:21:38 AM
Poisonwood	PW	NE	-81.387861	24.701565	5/16/2000	10:29:07 AM

BIG PINE FIRE ECOLOGY STUDY

Vascular plant species

Reference Codes: (1) Wunderlin, Richard P. 1998. Guide to the Vascular Plants of Florida. University Press of Florida, Gainesville. 806 pp.; (2) Long, Robert W. and Olga Lakela. 1976. A Flora of Tropical Florida: A Manual of the Seed Plants and Ferns of Southern Peninsular Florida. Banyan Books, Miami. 962 pp. ; (3) Correll, Donovan S. and Helen B. Correll. 1982 (Reprint 1996). Flora of the Bahama Archipelago. A.R.G. Gantner Verlag KG, Vaduz, Liechtenstein. 1692 pp.; and (4) Lellinger, David B. 1985. A Field Manual of the Ferns & Fern-allies of the United States & Canada. Smithsonian Institution Press, Washington, D.C. 389 pp.

Purple highlighting indicates species not occurring in subplots but observed in stands.

CLASS	FAMILY	SP CODE	SCIENTIFIC NAME	AUTHOR CITATION	REF	ALTERNATE NAME
PTERIDOPHYTE	PTERIDACEAE	ACRDAN	Acrostichum danaeifolium	Langsd. & Fisch.	1,4	
PTERIDOPHYTE	SCHIZAEACEAE	ANEADI	Anemia adiantifolia	(L.) Sw.	1,4	
PTERIDOPHYTE	PSILOTACEAE	PSINUD	Psilotum nudum	(L.) Beauv.	1,4	
PTERIDOPHYTE	DENNSTAEDTIACEAE	PTEAQU	Pteridium aquilinum	(L.) Kuhn	1	
PTERIDOPHYTE	PTERIDACEAE	PTEBAH	Pteris bahamensis	(J. Agardh) Fee	1,4	
PTERIDOPHYTE	THELYPTERIDACEAE	THEKUN	Thelypteris kunthii	(Desv.) C.V. Morton	1,4	
GYMNOSPERM	PINACEAE	PINELL	Pinus elliottii	Engelm.	1	
MONOCOT	CYPERACEAE	ABIOVA	Abildgaardia ovata	(Burm. F.) Kral	1	
MONOCOT	AGAVACEAE	AGADEC	Agave decipiens	Baker	1	
MONOCOT	LILIACEAE	ALEBRA	Aletris bracteata	Northr.	1	
MONOCOT	POACEAE	ANDCAB	Andropogon cabanisii	Hack.	2	1A. ternarius Michx.
MONOCOT	POACEAE	ANDGLO	Andropogon glomeratus	(Walter) Britton et al.	1	
MONOCOT	POACEAE	ANDVIR	Andropogon virginicus	L.	1	
MONOCOT	POACEAE	ARIPUR	Aristida purpurascens	Poir.	1	
MONOCOT	ORCHIDACEAE	BASCOR	Basiphyllaea corallicola	(Small) Ames	1	
MONOCOT	ORCHIDACEAE	BLEPUR	Bletia purpurea	(Lam.) DC.	1	
MONOCOT	POACEAE	CENECH	Cenchrus echinatus	L.	1	
MONOCOT	CYPERACEAE	CLAJAM	Cladium jamaicense	Crantz	1	
MONOCOT	ARECACEAE	COCARG	Coccothrinax argentata	(Jacq.) L.H. Bailey	1	
MONOCOT	POACEAE	DICDIC	Dichanthelium dichotomum	(L.) Gould	1	

CLASS	FAMILY	SP CODE	SCIENTIFIC NAME	AUTHOR CITATION	REF	ALTERNATE NAME
MONOCOT	CYPERACEAE	DICFLO	Dichromena floridensis	Britton ex Small	2	1Rhynchospora floridensis (Britton ex Small) H. Pfeiff.
MONOCOT	CYPERACEAE	ELECAR	Eleocharis caribaea	Torr.	1	
MONOCOT	POACEAE	ERAELL	Eragrostis elliottii	S. Watson	1	
MONOCOT	POACEAE	EUSPET	Eustachys petraea	(Sw.) Desv.	1	
MONOCOT	CYPERACEAE	FIMCAS	Fimbristylis castanea	(Michx.) Vahl	2	1F. spadicea (L.) Vahl
MONOCOT	CYPERACEAE	FIMSPA	Fimbristylis spathacea	Roth.	2	1F. cymosa R. Br.
MONOCOT	ORCHIDACEAE	HABQUI	Habenaria quinqueseta	(Michx.) Eaton	1	
MONOCOT	HYPOXIDACEAE	HYPWRI	Hypoxis wrightii	(Baker) Brackett	1	
MONOCOT	POACEAE	MUHFIL	Muhlenbergia filipes	M.A. Curtis	2	1M. capillaris (Lam.) Trin. var. filipes (M.A. Curtis) Chapm. ex Beal
MONOCOT	POACEAE	PANPOR	Panicum portoricense	Desv. ex Ham.	2	1Dicanthelium portoricense (Desv. ex Ham.) B.F. Hansen & Wunderlin
MONOCOT	POACEAE	PANRIG	Panicum rigidulum	Bosc ex Nees	1	
MONOCOT	POACEAE	PANVIR	Panicum virgatum	L.	1	
MONOCOT	POACEAE	PASCAE	Paspalum caespitosum	Flugge	1	
MONOCOT	POACEAE	PASMON	Paspalum monostachyum	Vasey ex Chapm.	1	
MONOCOT	POACEAE	PASSET	Paspalum setaceum	Michx.	1	
MONOCOT	CYPERACEAE	RHYMIC	Rhynchospora microcarpa	Baldwin ex A. Gray	1	
MONOCOT	CYPERACEAE	RHYODO	Rhynchospora odorata	C. Wright ex Griseb.	1	
MONOCOT	ARECACEAE	SABPAL	Sabal palmetto	(Walter) Lodd. ex Schult. & Schult. f.	1	
MONOCOT	ALISMATACEAE	SAGLAN	Sagittaria lancifolia subsp. lancifolia	L.	1	
MONOCOT	POACEAE	SCHGRA	Schizachyrium gracile	(Spreng.) Nash	1	
MONOCOT	POACEAE	SCHRHI	Schizachyrium rhizomatum	(Swallen) Gould	2	1S. scoparium var. scoparium (Michx.) Nash
MONOCOT	POACEAE	SCHSEM	Schizachyrium semiberbe	Nees	2	1S. sanguineum (Retz.) Alston
MONOCOT	CYPERACEAE	SCHNIG	Schoenus nigricans	L.	1	
MONOCOT	CYPERACEAE	SCLVER	Scleria verticillata	Muhl. Ex Willd.	1	
MONOCOT	ARECACEAE	SERREP	Serenoa repens	(W. Bartram) Small	1	

CLASS	FAMILY	SP CODE	SCIENTIFIC NAME	AUTHOR CITATION	REF	ALTERNATE NAME
MONOCOT	IRIDACEAE	SISMIA	Sisyrinchium miamiense	E.P. Bicknell		1S. angustifolium Mill.,2S. atlanticum E.P. Bicknell
MONOCOT	SMILACACEAE	SMIHAV	Smilax havanensis	Jacq.	1	
MONOCOT	POACEAE	SORSEC	Sorghastrum secundum	(Elliott) Nash	1	
MONOCOT	ORCHIDACEAE	SPITOR	Spiranthes torta	(Thunb.) Garay & H.R. Sweet	1	
MONOCOT	POACEAE	SPODOM	Sporobolus domingensis	(Trin.) Kunth	1	
MONOCOT	ARECACEAE	THRMOR	Thrinax morrisii	H. Wendl.	1	
MONOCOT	BROMELIACEAE	TILCIR	Tillandsia circinata	Schltdl.	2	1T. paucifolia Baker
MONOCOT	BROMELIACEAE	TILFLE	Tillandsia flexuosa	Sw.	1	
MONOCOT	BROMELIACEAE	TILUTR	Tillandsia utriculata	L.	1	
MONOCOT	BROMELIACEAE	TILVAL	Tillandsia valenzuelana	Rich.	2	1T. variabilis Schltdl.
MONOCOT	POACEAE	TRIFLO	Tripsacum floridanum	Porter ex Vasey	1	
MONOCOT	TYPHACEAE	TYPDOM	Typha domingensis	Pers.	1	
MONOCOT	ORCHIDACEAE	VANBAR	Vanilla barbellata	Rchb. f.	1	
DICOT	FABACEAE	ACAPIN	Acacia pinetorum	F.J. Herm.	1	
DICOT	EUPHORBIACEAE	ACACHA	Acalypha chamaedrifolia	(Lam.) Mull. Arg.	1	
DICOT	SCROPHULARIACEAE	AGAKEY	Agalinis keyensis	Pennell		1A. fasciculata (Ell.) Raf.,2A. filifolia (Nutt.) Raf.
DICOT	SCROPHULARIACEAE	AGAMAR	Agalinis maritima	(Raf.) Raf.	1	
DICOT	APOCYNACEAE	ANGBER	Angadenia berteri	(A. DC.) Miers	1	
DICOT	ANNONACEAE	ANNGLA	Annona glabra	L.	1	
DICOT	EUPHORBIACEAE	ARGBLO	Argythamnia blodgettii	(Torr. ex Chapm.) Chapm.	1	
DICOT	ASCLEPIADACEAE	ASCVIR	Asclepias viridis	Walter	1	
DICOT	ASTERACEAE	ASTDUM	Aster dumosus	L.	1	
DICOT	ASTERACEAE	ASTTEN	Aster tenuifolius	L.	1	
DICOT	AVICENNIACEAE	AVIGER	Avicennia germinans	(L.) L.	1	
DICOT	SCROPHULARIACEAE	BACMON	Bacopa monnieri	(L.) Pennell	1	
DICOT	RUBIACEAE	BORTER	Borreria terminalis	Small	2	1Spermacoce terminalis (Small) Kartesz & Gandhi

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DICOT	BORAGINACEAE	BOUCASS	Bourreria cassinifolia	(A. Rich.) Griseb.	1	
DICOT	SCROPHULARIACEAE	BUCFLO	Buchnera floridana	Gand.	2	1B. americana L.
DICOT	SAPOTACEAE	BUMCEL	Bumelia celastrina	Kunth	2	1Sideroxylon celastrinum (Kunth) T.D. Penn.
DICOT	SAPOTACEAE	BUMSAL	Bumelia salicifolia	(L.) Sw.	3	1Sideroxylon salicifolium (L.) Lam.,2Dipholis salicifolia (L.) A. DC.
DICOT	BURSERACEAE	BURSIM	Bursera simaruba	(L.) Sarg.	1	
DICOT	MALPIGHIACEAE	BYRLUC	Byrsonima lucida	(Mill.) DC.	1	
DICOT	FABACEAE	CAEPAU	Caesalpinia pauciflora	(Griseb.) C. Wright	1	
DICOT	SCROPHULARIACEAE	CAPBIF	Capraria biflora	L.	1	
DICOT	FABACEAE	CASBAH	Cassia bahamensis	Mill.	2	1Senna mexicana (Jacq.) H.S. Irwin & Barneby
DICOT	LAURACEAE	CASFIL	Cassytha filiformis	L.	1	
DICOT	CASUARINACEAE	CASEQU	Casuarina equisetifolia	L.	1	
DICOT	RUBIACEAE	CATPAR	Catesbaea parviflora	Sw.	1	
DICOT	APIACEAE	CENASI	Centella asiatica	(L.) Urb.	1	
DICOT	FABACEAE	CENVIR	Centrosema virginianum	(L.) Benth.	1	
DICOT	FABACEAE	CHAASP	Chamaecrista aspera	(Muhl. ex Elliott) Greene		1C. nictitans (L.) Moench var. aspera (Muhl. ex Elliott) H.S. Irwin & Barneby,2Cassia aspera Muhl.
DICOT	FABACEAE	CHALIN	Chamaecrista lineata	(Sw.) Greene	1	
DICOT	EUPHORBIACEAE	CHAADE	Chamaesyce adenoptera subsp. pergamena	(Bertol.) Small/(Small) Burch	2	1C. pergamena (Small) Small
DICOT	EUPHORBIACEAE	CHADEL	Chamaesyce deltoidea	(Engelm. ex Chapm.) Small	1	
DICOT	EUPHORBIACEAE	CHAPOR	Chamaesyce porteriana	Small	1	
DICOT	ASTERACEAE	CHADEN	Chaptalia dentata	(L.) Cassini	2	
DICOT	RUBIACEAE	CHIALB	Chiococca alba	(L.) Hitchc.	1	
DICOT	RUBIACEAE	CHIPIN	Chiococca pinetorum	Britton	2	1C. alba (L.) Hitchc., 3C.parvifolia Wullschl. ex Griseb.
DICOT	CHRYSOBALANACEAE	CHRICA	Chrysobalanus icaco	L.	1	

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DICOT	SAPOTACEAE	CHROLI	Chrysophyllum oliviforme	L.	1	
DICOT	ASTERACEAE	CIRHOR	Cirsium horridulum	Michx.	1	
DICOT	EUPHORBIACEAE	CNISTI	Cnidoscolus stimulosus	(Michx.) Engelm. & A. Gray	1	
DICOT	POLYGONACEAE	COCDIV	Coccoloba diversifolia	Jacq.	1	
DICOT	POLYGONACEAE	COCUVI	Coccoloba uvifera	(L.) L.	1	
DICOT	COMBRETACEAE	CONERE	Conocarpus erectus	L.	1	
DICOT	ASTERACEAE	CORLEA	Coreopsis leavenworthii	Torr. & A. Gray	1	
DICOT	CELASTRACEAE	CROILI	Crossopetalum ilicifolium	(Por.) Kuntze	1	
DICOT	CELASTRACEAE	CRORHA	Crossopetalum rhacoma	Crantz	1	
DICOT	FABACEAE	CROMAR	Crotalaria maritima	Chapm.	2	1C. rotundifolia J.F. Gmel.
DICOT	EUPHORBIACEAE	CROGLA	Croton glandulosus	L.	1	
DICOT	EUPHORBIACEAE	CROLIN	Croton linearis	Jacq.	1	
DICOT	ASCLEPIADACEAE	CYNBLO	Cynanchum blodgettii	(A. Gray) Shinners	1	
DICOT	LOGANIACEAE	CYNSES	Cynoctonum sessilifolium	J.F. Gmel.	2	1Mitreola sessilifolia (J.F. Gmel.) G. Don
DICOT	EBENACEAE	DIOVIR	Diospyros virginiana	L.	1	
DICOT	ACANTHACEAE	DYSOBL	Dyschoriste oblongifolia	(Michx.) Kuntze	1	
DICOT	APOCYNACEAE	ECHUMB	Echites umbellata	Jacq.	1	
DICOT	RUBIACEAE	ERIFRU	Erithalis fruticosa	L.	1	
DICOT	RUBIACEAE	ERNLIT	Ernodea littoralis	Sw.	1	
DICOT	MYRTACEAE	EUGAXI	Eugenia axillaris	(Sw.) Willd.	1	
DICOT	MYRTACEAE	EUGFOE	Eugenia foetida	Pers.	1	
DICOT	GENTIANACEAE	EUSEXA	Eustoma exaltatum	(L.) Salisb. ex G. Don	1	
DICOT	CONVOLVULACEAE	EVOGRI	Evolvulus grisebachii	Peter	1	
DICOT	CONVOLVULACEAE	EVOSER	Evolvulus sericeus	Sw.	1	
DICOT	MORACEAE	FICAUR	Ficus aurea	Nutt.	1	
DICOT	MORACEAE	FICCIT	Ficus citrifolia	Mill.	1	
DICOT	ASTERACEAE	FLALIN	Flaveria linearis	Lag.	1	
DICOT	FABACEAE	GALPAR	Galactia parvifolia	A. Rich.	2	1C. volubilis (L.) Britton
DICOT	NYCTAGINACEAE	GUADIS	Guapira discolor	(Spreng.) Little	1	
DICOT	RUBIACEAE	GUESCA	Guettarda scabra	(L.) Vent.	1	

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DICOT	RUBIACEAE	HEDNIG	Hedyotis nigricans	(Lam.) Fosberg	1	
DICOT	BORAGINACEAE	HELPOL	Heliotropium polyphyllum	Lehm.	1	
DICOT	ASTERACEAE	HETGRA	Heterotheca graminifolia	(Michx.) Shinners	2	1Pityopsis graminifolia (Michx.) Nutt.
DICOT	FABACEAE	INDMIN	Indigofera miniata	Ortega	1	
DICOT	CONVOLVULACEAE	IPOMOEA	Ipomoea			
DICOT	CONVOLVULACEAE	JACPEN	Jacquemontia pentanthos	(Jacq.) G. Don	1	
DICOT	THEOPHRASTACEAE	JACKEY	Jacquinia keyensis	Mez	1	
DICOT	VERBENACEAE	LANINV	Lantana involucrata	L.	1	
DICOT	FABACEAE	LEULEU	Leucaena leucocephala	(Lam.) de Wit	1	
DICOT	ASTERACEAE	LIAGRA	Liatris gracilis	Pursh	1	
DICOT	ASTERACEAE	LIALAE	Liatris laevigata	Nutt.		1L. tenuifolia Nutt. var. quadriflora Chapm.,2L. tenuifolia var. laevigata (Nutt.) B.L. Rob.
DICOT	CHRYSOBALANACEAE	LICMIC	Licania michauxii	Prance	1	
DICOT	LINACEAE	LINARE	Linum arenicola	(Small) H.J.P. Winkler	1	
DICOT	CAMPANULACEAE	LOBGLA	Lobelia glandulosa	Walt.	1	
DICOT	ONAGRACEAE	LUDMIC	Ludwigia microcarpa	Michx.	1	
DICOT	SAPOTACEAE	MANBAH	Manilkara bahamensis	(Baker) H.J. Lam & B. Meeuse	2	1M. jaimiqui (C. Wright ex Griseb.) Dubard subsp. emarginata (L.) Cronquist
DICOT	ASTERACEAE	MELPAR	Melanthera parvifolia	Small	2	1M. nivea (L.) Small
DICOT	ANACARDIACEAE	METTOX	Metopium toxiferum	(L.) Krug & Urb.	1	
DICOT	ASTERACEAE	MIKSCA	Mikania scandens	(L.) Willd.	1	
DICOT	RUBIACEAE	MORROY	Morinda royoc	L.	1	
DICOT	MYRICACEAE	MYRCER	Myrica cerifera	L.	1	
DICOT	MYRSINACEAE	MYRFLO	Myrsine floridana	A. DC.	3	1Rapanea punctata (Lam.) Lund.,2M. guianensis (Aubl.) Kuntze
DICOT	FABACEAE	NEPPUB	Neptunia pubescens	Benth.	1	
DICOT	CACTACEAE	OPUSTR	Opuntia stricta	(Haw.) Haw.	1	
DICOT	PASSIFLORACEAE	PASSUB	Passiflora suberosa	L.	1	
DICOT	ASTERACEAE	PECLEP	Pectis leptocephala	(Cass.) Urb.	2	1P. glaucescens (Cass.) D.J. Keil

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DICOT	EUPHORBIACEAE	PHYPEN	Phyllanthus pentaphyllus	C. Wright ex Griseb.	1	
DICOT	SOLANACEAE	PHYANG	Physalis angustifolia	Nutt.	1	
DICOT	SOLANACEAE	PHYARE	Physalis arenicola	Kearney	1	
DICOT	SOLANACEAE	PHYVIS	Physalis viscosa	L.	2	1P. walteri Nutt.
DICOT	LENTIBULARIACEAE	PINPUM	Pinguicula pumila	Michx.	1	
DICOT	TURNERACEAE	PIRCAR	Piriqueta caroliniana	(Walter) Urb.	1	
DICOT	FABACEAE	PISPIS	Piscidia piscipula	(L.) Sarg.	1	
DICOT	NYCTAGINACEAE	PISROT	Pisonia rotundata	Griseb.	1	
DICOT	FABACEAE	PITGUA	Pithecellobium guadalupense	(Pers.) Chapm.		1,2P. keyense Britton ex Britton & Rose
DICOT	ASTERACEAE	PLUROS	Pluchea rosea	R.K. Godfrey	1	
DICOT	EUPHORBIACEAE	POIPIN	Poinsettia pinetorum	Small	1	
DICOT	POLYGALACEAE	POLBOY	Polygala boykinii	Nutt.	1	
DICOT	POLYGALACEAE	POLGRA	Polygala grandiflora	Walter	1	
DICOT	MYRTACEAE	PSILON	Psidium longipes	(O. Berg.) McVaugh	1	
DICOT	ASTERACEAE	PTEPYC	Pterocaulon pycnostachyum	(Michx.) Elliott	1	
DICOT	RUBIACEAE	RANACU	Randia aculeata	L.	1	
DICOT	RHAMNACEAE	REYSEP	Reynosia septentrionalis	Urb.	1	
DICOT	RHIZOPHORACEAE	RHIMAN	Rhizophora mangle	l.	1	
DICOT	ANACARDIACEAE	RHURAD	Rhus radicans	L.	3	1,2Toxicodendron radicans (L.) Kuntze
DICOT	FABACEAE	RHYPAR	Rhynchosia parvifolia	DC.	1	
DICOT	ACANTHACEAE	RUECAR	Ruellia caroliniensis	(J.F. Gmel.) Steud.	1	
DICOT	GENTIANACEAE	SABSTE	Sabatia stellaris	Pursh	1	
DICOT	ASTERACEAE	SACPOL	Sachsia polycephala	Griseb.	1	
DICOT	PRIMULACEAE	SAMEBR	Samolus ebracteatus	Kunth	1	
DICOT	ANACARDIACEAE	SCHTER	Schinus terebinthifolius	Raddi	1	
DICOT	LAMIACEAE	SCUHAV	Scutellaria havanensis	Jacq.	1	
DICOT	MALVACEAE	SIDELL	Sida elliottii	Torr. & A. Gray	1	
DICOT	SIMAROUBACEAE	SIMGLA	Simarouba glauca	DC.	1	
DICOT	SOLANACEAE	SOLBAH	Solanum bahamense	L.	1	
DICOT	SOLANACEAE	SOLDON	Solanum donianum	Walp.	3	1S. verbascifolium L.

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DICOT	SOLANACEAE	SOLERI	Solanum erianthum	D. Don	1	
DICOT	ASTERACEAE	SOLSTR	Solidago stricta	Aiton	1	
DICOT	FABACEAE	SOPTOM	Sophora tomentosa	L.	1	
DICOT	RUBIACEAE	STRMAR	Strumpfia maritima	Jacq.	1	
DICOT	FABACEAE	STYHAM	Stylosanthes hamata	(L.) Taub.	1	
DICOT	MELIACEAE	SWIMAH	Swietenia mahagoni	(L.) Jacq.	1	
DICOT	EUPHORBIACEAE	TRASAX	Tragia saxicola	Small	1	
DICOT	APOCYNACEAE	URELUT	Urechites lutea	(L.) Britton	2	1Pentalinon luteum (L.) B.F. Hansen & Wunderlin
DICOT	ASTERACEAE	VERBLO	Vernonia blodgettii	Small	1	
DICOT	OLACACEAE	XIMAME	Ximenia americana	L.	1	